INVESTIGATING PREFERENCE HETEROGENEITY FOR RESTORING ESTUARINE ECOSYSTEM SERVICES

Valeria Maria Toledo-Gallegos

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Investigating preference heterogeneity for restoring estuarine ecosystem services

Valeria Maria Toledo-Gallegos

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Abstract

Given the general decline of estuarine ecosystem services (ES), policy makers require to understand further the drivers and barriers to increase society's support for policies restoring them. The objective of this study is to identify significant sources of preference heterogeneity for improvements in flood control, recreation and biodiversity levels, resulting from tax-funded restoration projects that would be developed in the Clyde, Forth, and Tay catchment.

We used data from a discrete choice experiment conducted in Scotland and applied several choice modelling techniques (e.g. MNL, RPL, HMXL, posterior analysis) to explore the effect of respondents socioeconomic characteristics, their latent attitudes and the local geographical context on their preferences towards policies managing estuarine ES.

We found a positive and significant willingness to pay (WTP) for improving all ES, although differences in WTP estimates exist for all estuarine ES, across catchments and between user types. Recreation values were found to be lower on average than either flood control or biodiversity conservation, while preference differences emerge due to whether people live within a catchment and whether they visit it for recreational purposes or not. People visiting the areas for doing outdoor activities presented a higher latent environmental consciousness attitude. Moreover, environmentally conscious individuals showed stronger preferences for management alternatives delivering estuarine ES improvements. Finally, the presence of significant local clusters of WTP estimates suggests that respondents' preferences interact with their immediate spatial context. Nonetheless, the local clusters of WTP for estuarine ES improvements are distributed similarly in space regardless of the ES in question, or the estuary under consideration.

The research findings can be informative for designing more efficient and contextualised policies. Moreover, they can be helpful in raising the social acceptability of the policies aiming to manage estuarine ES in Scotland.

General acknowledgements

This PhD represents much more than an academic achievement to me, since developing this research has been a continuous process of learning and development in both the academic and personal sense.

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This thesis is dedicated to my mom, who has always been an inspiring figure in my life, and to my sister, who has always supported me unconditionally.

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List of acronyms

Access and benefit sharing	**ABS**
Akaike information criterion	**AIC**
Alternative specific constant	**ASC**
Bayesian information criterion	**BIC**
Cost-benefit analysis	**CBA**
Confidence intervals	**CI**
Choice modelling	**CM**
Complete spatial randomness and independence	**CSRI**
Contingent valuation	**CV**
Discrete choice experiment	**DCE**
Ecologically conscious consumer behaviour	**ECCB**
Ecosystem services	**ES**
European Union	**EU**
British pound sterling	**GBP**
Geographical information system	**GIS**
Hybrid choice model	**HCM**
High-High (hotspot)	**HH**
High-Low	**HL**
Hybrid latent class	**HLC**
Hybrid multinomial logit	**HMNL**
Hybrid mixed logit	**HMXL**
Integrated Choice and Latent Variable	**ICLV**
Independence of irrelevant alternatives	**IIA**
Independent and identically distributed	**IID**
Internet protocol	**IP**
Latent class model	**LCM**
Low-High	**LH**
Local indicator of spatial autocorrelation	**LISA**
Log-likelihood	**LL**
Low-Low (coldspot)	**LL**
Multiple Indicator Multiple Cause	**MIMIC**
Multinomial logit model	**MNL**
Not applicable	**NA**
New Ecological Paradigm	**NEP**
Not significant	**NS**
Pro-environmental behaviour	**PEB**
Payments for ecosystem services	**PES**
Revealed preferences	**RP**
Random parameter logit model	**RPL**
Random utility maximisation	**RUM**
Structural Equation Models	**SEM**
Scottish Environment Protection Agency	**SEPA**

Stated preferences	**SP**
The Economics of Ecosystems and Biodiversity	**TEEB**
Total Economic Value	**TEV**
Theory of Planned Behaviour	**TPB**
United Kingdom	**UK**
United States of America	**USA**
University Teaching and Research Ethics Committee	**UTREC**
Willingness to accept	**WTA**
Willingness to donate	**WTD**
Willingness to pay	**WTP**

Part I. Introduction, background and policy setting

Chapter 1. General introduction and study area

This research uses the ecosystem services (ES) approach as the conceptual framework to connect the estuarine natural environment with society through the provision of benefits. The work presented in this thesis originates from an interest in generating more effective and contextualised environmental policies that aim to tackle the escalating degradation of estuarine ES.

This chapter introduces the essential conceptual foundations of the study (section 1.1) and provides some contextual background for the research (section 1.2). Afterwards, this chapter outlines the research objectives and motivation (section 1.3 and 1.4), and finally describes the general thesis layout and research outputs in section 1.5.

1.1. Conceptual foundations

1.1.1. Ecosystem services and wellbeing

Ecosystem services are the direct or indirect benefits that humanity obtains from natural ecosystems (Millenium Ecosystems Assessment, 2005). The term ‘ecosystem services’ has become an increasingly popular concept for researchers and policy makers as it allows one to link society with nature through the concept of wellbeing. Some authors have claimed that the concept of ES has an ‘anthropocentric’ nature which leads to the ‘commodification of nature’ (Gómez-Baggethun and Ruiz-Pérez, 2011; Schröter et al., 2014) and the danger of acting as a ‘complexity blinder’(Norgaard, 2010). However, the broad acceptance of the ES framework does not only rely on its instrumental utility for increasing the awareness of societal dependence on nature. The use of the ES concept has further advantages, such as its capacity of acting as a ‘metaphor‘ or ‘common language’ that promotes interdisciplinary science (Hoppe, 2011), as well as its function as a ‘theoretical platform’ for different stakeholders to join research efforts towards environmental conservation. Finally, the ES concept has played a vital role in promoting sustainable management and conservation actions (Luck et al., 2012). For instance, the development of national (UK National Ecosystem Assessment, 2011) and global ES assessments (Millenium Ecosystems Assessment, 2005; Natural Capital Project, 2012;

TEEB, 2010; The World Bank, 2015; United Nations, 2012) have raised attention to the urgency of acting towards solving environmental problems, as well as facilitating the integration of natural capital into the policy framework through the use of methods that measure and value their stocks and flows.

A growing body of literature suggests that the capacity of nature for providing benefits to society is intimately linked to the functioning and the biophysical structure of ecosystems (Balvanera et al., 2006; Cardinale et al., 2012). Since the 1950s there has been a general trend of decline and degradation of ES, with approximately 60% of the ES being used unsustainably worldwide (Millenium Ecosystems Assessment, 2005), and 66% in Europe (Bourguignon, 2017). Certainly, technological and knowledge advances might act as a buffer against environmental change. However, in the context of an accelerating environmental degradation coupled with the limited substitutability of the functions provided by natural capital (Costanza et al., 1997; Ekins et al., 2003), this problem becomes central.

From an economic perspective, externalities exist when individual consumption or production affect positively or negatively the consumption or production of others (Buchanan and Stubblebine, 1962). Environmental degradation and pollution is an example of a negative externality, whereas ES provision is considered a positive externality. Polluting estuaries, for instance, could have repercussions on the fishing and tourism industry and a harmful effect on society through the loss of income or the rise of health expenses. On the other hand, if farmers within the estuary catchment adopt sustainable management that enhances the provision of ES, society can benefit from enjoying a more beautiful and safe environment. Therefore, if the aim is to maintain the wide range of benefits society derives from the ecosystems (e.g. health, wellbeing, poverty alleviation), research efforts may focus on finding ways to reduce human-induced pressures to nature and to restore nature's functioning.

1.2. Research background

1.2.1. Degradation of estuarine ecosystem services

Estuaries are biomes existing on the transitional zone linking rivers and wetlands with marine and oceanic habitats (Basset et al., 2013). These transitional ecosystems present a complex ecological functioning (Jacobs et al., 2014) and have dynamic geomorphology (McLusky, 1981) which is continuously influenced by the physical, chemical or biological processes operating in the contiguous terrestrial and marine ecosystems. As a result of the constant exposure to naturally stressful conditions, estuarine biological communities tend to have a low number of species, but high abundance (Elliott and Quintino, 2007).

Despite their physical and biological dynamism, high estuarine productivity underpins the provision of a wide range of ES and makes them economically and socially valuable ecosystems. Currently, there are three international systems used to classify ES according to the type of benefit they provide to society, which are developed by the Millenium Ecosystems Assessment (2005), the Economics of Ecosystems and Biodiversity (TEEB, 2010) and the International Classification of Ecosystem Services (Haines-Young and Potschin, 2013). This study uses the TEEB system as it is a classification system extensively used in valuation studies across Europe. In accordance with the classification used by 'The Economics of Ecosystems and Biodiversity' (TEEB) described in De Groot et al. (2010), ecosystems provide four types of ES: i) provisioning services, ii) regulating services, iii) habitat or supporting services, and iv) cultural services. The work of Jacobs et al. (2014) identified a total of 46 estuarine ES of which 33% are classified as provisioning, 54% regulating, 11% cultural and 2% supporting services. A detailed classification of estuarine ecosystem services within the TEEB framework can be found in annex 1. Some examples of estuarine provisioning services include the provision of food (e.g. fish), raw materials for construction and fuel; as well as the provision of water for navigating, household and industrial use. Estuarine regulating services refers to their capacity to sequestrate carbon; protect people from floods, storms and other extreme events through the regulation of the flood water storage and draining river water, reducing waves and dissipating tidal and river energy. Regarding cultural services, the natural environment of estuaries is a space for recreation, aesthetical enjoyment and inspiration for culture, art or design. Finally, the habitat or supporting services relate to the estuarine capacity of supporting all previous ES by maintaining biodiversity and keeping genetic diversity, as well as providing habitat for aquatic and terrestrial species.

Not surprisingly, humankind has historically benefited from estuarine ecosystems and have often located their settlements near estuaries. Historical developments in European coastal and estuarine ecosystems have resulted in the reduction of their original surface area by 50% (Airoldi and Beck, 2007). In agreement with the definition by Davidson et al. (1991), it is possible to identify approximately 160 estuaries in the United Kingdom[1] (UK) which represent a quarter of the estuaries located in northwest Europe (Robins et al., 2016a). Austin et al. (2000) report that by the year of 2000 two thirds of the human population in the UK lived near estuaries affecting the environmental quality of 88% of the British estuaries.

Estuaries are one of the most relevant ecological features forming the coastal landscape. This is not only because they are considered one of the most biologically productive biomes in the world (Day et al., 2012), but also because of the role they have on sustaining human civilisation, which has historically benefited from the variety of ES that estuaries provide. Nonetheless, estuaries rank among the most heavily damaged and exploited ecosystems worldwide (Kennish, 2002; Worm et al., 2006).

The escalating impacts that come with population growth and coastal development undermine estuarine ecosystems resilience (Lotze et al., 2006) and could accelerate the occurrence of changes in its structure and functionality. A study by Pinto et al. (2013) shows that estuarine habitat conversion which leads to heterogeneous resource distribution and a decrease in patch connectivity (i.e. reduction on the area of structured habitats) impacts the ecosystem stability and subsequently lead to the loss of ES. Thus the levels of ES provision are intimately linked with the estuarine ecosystem management and protection practices.

In the UK, changes in the biophysical environment of estuaries are due to anthropogenic activities and forces of climate change (Robins et al., 2016b). Some examples of economic activities threatening estuarine health are aquaculture and shellfisheries; oil, gas and electric power production; intensive farming and agriculture; forestation or deforestation; urban development and tourism; coastal management, dredging and filling

[1] See 'The estuary guide' for details on UK estuarine habitat and characterization (Defra/EA Flood and Coastal Erosion Risk Management R&D Programme, 2008)

(Kennish, 2002; Lee et al., 2006; Robins et al., 2016a). Their development has led to environmental problems caused by the excessive nutrient concentration, over-fishing, water chemical pollution, freshwater diversion, the introduction of invasive species and coastal erosion (Kennish, 2002; Nehring, 2006; Statham, 2012). More recently these issues are being coupled with the exposure and loss of habitat resulting from climate change (Robins et al., 2016a).

Policies that aim to restore or enhance the provision levels of estuarine ES require further understanding of their interdependence expressed in positive (synergies) or negative (trade-offs) relationships (Bennett et al., 2009; Jacobs et al., 2014). In doing so, research should acknowledge the complexity of the estuarine ES links and study these in ‘bundles’. The knowledge derived from such studies is essential for the design of sustainable management plans which seek to maximise existing synergies (Maes et al., 2012).

Several authors advocate generating environmental policies which use strategies that simultaneously enhance multiple ES (Egoh et al., 2008; Gordon et al., 2008; Jackson et al., 2013; Nelson et al., 2009). The present study takes into consideration these potential linkages and studies three policy-relevant estuarine ES: i) flood risk control, ii) biodiversity, and iii) recreation. Table 1-1 describes the names of the ES and the category given by the TEEB classification (de Groot et al., 2010).

Table 1-1 Ecosystem services name and TEEB classification

Ecosystem service	TEEB name	TEEB category
Flood control	Regulation of water flows	Regulating services
Biodiversity	Maintenance of genetic diversity	Habitat or supporting services
Recreation	Opportunities for recreation and tourism	Cultural services

We selected estuarine ES which are relevant for the contemporary management of estuaries because of the following reasons. First, there is evidence supporting a rise in sea level and river flows in the UK as a consequence of climate change which has the potential of increasing the flood risk in estuarine ecosystems (Robins et al., 2016b). Second, there is a growing body of literature that indicates that biodiverse estuarine ecosystems are necessary to underpin the provision of the rest of the ES (Balvanera et al., 2006; Hector and Bagchi, 2007). Finally, considering cultural ES such as recreation is relevant as they play a crucial role in strengthening the connection of society with nature and act as an incentive to engage in environmental conservation and protection measures (Daniel et al., 2012).

The design of management plans for estuarine ecosystems is particularly challenging due to their dynamic and complex ecological functioning; and because the consequences of their management could affect a significant proportion of the worldwide population (Granek et al., 2010). Literature evaluating management policies for estuarine ES has provided some recommendations for achieving more effective outcomes. First, it is recommended to use a *catchment scale* analysis (i.e. area of land which collects water heading to the estuary) which acknowledges that changes happening in terrestrial, riverine and coastal ecosystems could impact estuarine health (Kennish, 2002). Second, it is relevant to increase the insights on the effect of the management policy chosen not only for a specific service but to a ‘bundle’ of estuarine ES (De Groot et al., 2010). Finally, Jacobs et al. (2014) argue for the integration of knowledge regarding the supply and demand of estuarine ES with surveying and mapping methodologies.

This research takes into account all the previous suggestions and develops an analysis of estuarine ES from a value perspective. It uses an environmental valuation technique called discrete choice experiment (DCE) for gaining insights into society’s preferences and the drivers of demand, which are referred in the following chapters as the ‘sources of preference heterogeneity’.

1.2.2. Economic contributions of ecosystem services in Scotland

Although some human efforts are commonly required (e.g. energy inputs, labour, management, infrastructure, research and technology expenditures) in order for society to fully absorb the benefits from ecosystems (Maes et al., 2013). The ES are vital in Scotland due to their numerous direct and indirect contributions to human well-being and the considerable gains they generate for the economic system. The Scottish Government (2013) suggest that overall, ES represent a financial asset with an estimated value of £21.5-£23 billion per year.

The benefits provided by ecosystems impact Scottish citizens' well-being in different ways. The provisioning services, for instance, fulfil the physical and basic material needs of humans such as food, water, medicine, and energy sources while contributing to the maintenance of physical health. From the years 2004 to 2008 provisioning services in Scotland were estimated to have an average annual total direct value of £2.5 billion for the agriculture, forestry and marine fisheries industries (Green et al., 2011). That is excluding the total value of minerals produced onshore (£650 million) and the value of oil and gas exports which was estimated to be £2.4 billion in 2011 (Critchlow-Watton et al., 2014).

Cultural services are the non-physical benefits that society obtain from the environment which helps to achieve emotional health and socio-cultural development. The cultural benefits relate to the environment's capacity for providing spiritual enrichment, cognitive development, recreational and aesthetical experiences. One example of a cultural service for which national accounts exist is nature-based recreation. This type of recreational service was studied in the present research since it was estimated to provide Scotland with £1.4 billion in income associated with generated jobs (Bryden et al., 2010) and around £2.6 billion in expenditure for outdoor visits paid during the year 2012 (Wilson and Stewart, 2013).

Regulating services refer to the environmental processes necessary to fulfil the safety and security needs of our civilisation. They relate to an ecosystem's capacity to regulate their environmental quality (air, soil, water) and resilience to face environmental shocks (flood, erosion, pollution, diseases and natural hazards). Flood control is another selected ES because the economic cost of flooding in Scotland for the year 2003 averaged £31.5 million per year from inland flooding and £19.1 million from coastal flooding, and this cost was forecasted to be £52.9 million by 2050 (Werritty and Chatterton, 2003).

Finally, habitat and supporting services relate to the environmental cycles that help to maintain healthy and biodiverse ecosystem structures. Researching biodiversity is important for this research as we rely on it for providing shelter and producing all other ES. Invasive species directly threaten native species (impact biodiversity) and estuarine ecosystem functioning (Nehring, 2006; Williams and Grosholz, 2008). Developing strategies for controlling them is relevant as their current management represent a cost to the Scottish economy of £250 million per annum (Williams et al., 2010).

1.3. Research motivation

The degradation of estuarine ES has significant consequences for the economic development of Scotland, as they represent a direct source of wealth to society and they underpin vital economic sectors for the country such as agriculture, forestry and tourism. The mismanagement of estuarine ecosystems has resulted in the loss of ES which accentuates with the rise of the population in coastal UK. The continuous degradation of estuarine environmental quality could lead to overpassing critical natural stock depletion thresholds, also called ‘tipping points’. The crossing of estuarine tipping points could have irreversible consequences and might result in the abrupt decline or collapse of ES flows (Riche et al., 2014; Wang et al., 2015).

In order to maintain the flow of benefits that society obtains from estuarine ecosystems, it is therefore necessary to develop environmental policies which restore the ES provision levels. The development of environmental management plans which secure the ES flows coming from the Clyde, Forth, and Tay estuary (see section 3.1.1 for a description of their characteristics) is particularly important as their catchment areas contain the most densely populated settlements in Scotland, and their anthropogenic pressures are likely to increase with time.

In the real world, the development of environmental policies not only responds to environmental emergencies, but also responds to specific social and political circumstances. Environmental policies which consider public needs and preferences in the design and implementation of conservation measures are more likely to be effective and achieve the intended outcome successfully (Newig and Fritsch, 2009). Information regarding society’s environmental preferences might be used in unison with natural scientists’ recommendations to prioritise the conservation measures which need to be developed under limited policy budgets.

Considering society’s environmental preferences for managing ES represents a step forward in the generation of environmental policies which balance the urgency for restoring ES and the social demands for specific benefits. Thus, the study of preference heterogeneity is a useful tool for designing policies which reduce conflicts that arise from environmental management (Ojea & Loureiro, 2007) and to consider equity concerns (Abrell et al., 2016). Moreover, the recognition of the sources of preference heterogeneity provides decision makers with greater insight into the motives for and barriers to social support of restoration policies. Finally, it helps to identify what to target when seeking the adoption of environmentally responsible practices, which is vital for the current scenario of severe environmental degradation and resource depletion.

1.4. Research aims and objectives

Policies that aim to restore or enhance the provision of ES in estuarine ecosystems should consider both the ecological and social perspectives of their management. Understanding the biological structures that need to be restored for providing estuarine ES is out of the scope of this study and has been previously examined in the ecological body of literature (Boerema and Meire, 2017; Elliott et al., 2016; Grabowski and Peterson, 2007; Hengst et al., 2010).

Instead, this research aims to contribute to understanding the social perspective of environmental management by identifying significant drivers of environmental preferences. Given that the recognition of the variety of factors (e.g. socioeconomic, physiological, space and time) influencing individuals' processes of decision making is vital for developing more efficient and optimal environmental policies.

The present work explores whether studies valuing environmental goods and services have recognised the diversity of sources of preference heterogeneity (see literature review in chapter 2). In the above context, the *general aim of this research* is to identify the potential sources of environmental preference heterogeneity and to evaluate their effect on individuals' willingness to pay (WTP) for policies restoring ES provision in the Clyde, Forth, and Tay catchment areas. Within this broad aim, we formulated three specific study objectives and questions that are addressed individually by each empirical chapter, which are:

- *Specific objective 1:* To examine how the individuals' preferences for policies restoring estuarine ES are influenced by a study site characteristics and by their use characteristics.

  Questions derived from objective 1: Are the study site characteristics and user characteristics a significant source of preference heterogeneity? Do environmental preference vary across case studies with different environmental quality, and according to the degree to which users make direct use of the ES?
- *Specific objective 2:* To compare the geographical distribution of the environmental preferences (i.e. spatial preference heterogeneity) related to the three estuarine ES, as well as the three catchment areas.

  Questions derived from objective 2: Is the spatial context a significant source of preference heterogeneity? Are patterns of spatial preference heterogeneity constant across environmental goods?
- *Specific objective 3:* To assess the role of environmental attitudes (i.e. environmental consciousness) as an underlying source of preference heterogeneity of policies restoring estuarine ES.

  Question 3: Are latent attitudes towards the environment a significant source of preference heterogeneity? How do environmental attitudes impact individuals' support for policies restoring estuarine ES?

All research objectives contribute to the existing literature by deepening our understanding of sources of heterogeneity. However, we place particular attention on the effect of socioeconomic characteristics, latent attitudes and the geographical context.

S*pecific objective 1* augments the preference heterogeneity analysis and uses a comparative approach to understand the effect of both, external and contextual factors to individuals' decision making process. *Specific objective 2* contributes to the literature by attempting to understand the spatial patterns of environmental preference heterogeneity. This analysis consists of the first application questioning the existence of pattern similarities between different ES types and case studies. Finally, *specific objective 3* contributes to a recent trend in the literature which explores the effect of psycho-cognitive factors in environmental valuation and develops a novel application to the ES framework.

As the literature review in the chapter 2 shows, previous research has analysed the impact of these three factors on society's environmental preferences independently. However, there is a lack of effort in the environmental valuation literature to integrate these results; to generate policy recommendations that account for the combined effect of socioeconomic, spatial and attitudinal factors on society's preferences for estuarine ES when designing environmental restoration projects. Thus, to provide this combined analysis of preference heterogeneity is *the final aim* of this study.

1.5. Thesis overview and research outputs

This research applies an economic approach of analysis, in which individuals' preferences and their value judgments are expressed via their WTP for environmental improvements. We use data derived from a DCE exploring preferences for estuarine ES improvements in Scotland. Three empirical chapters develop different choice modelling techniques and aim to demonstrate how the socioeconomic, attitudinal and geographical contextual factors impact society's environmental preferences, respectively.

The structure and content of the remaining chapters are as follows. Chapter 2 summarises the theory underlying economic valuation in general and DCE specifically, as well as introduces the body of literature related to environmental preference heterogeneity. Importantly, chapter 2 reviews and characterises the current research applications exploring the effect of socioeconomic, attitudes and the spatial context on environmental preferences and values. Chapter 3 details the DCE design, as well as the survey and sampling design. This chapter also provides further explanation of the data preparation and validation process, to finally present a summary of the total sample characteristics. Chapters 4, 5 and 6 present empirical studies applying various choice modelling techniques to address the questions related to the *specific objectives 1, 2 and 3, respectively*. Finally, chapter 7 presents a synthesis of major findings and discusses their implications for environmental policy planning. Additionally, it discusses the empirical and methodological achievements and limitations, to then conclude with some suggestions for future work.

The research outputs from the empirical chapters have been presented in several conferences including the in Nineteenth Annual International BIOECON conference; the

European Society for Ecological Economics Conference (ESEE); the Scottish Ecology, Environment & Conservation Conference; Envecon 2017 and 2018: Applied Environmental Economics Conference for presenting doctoral research findings; as well as the World Congress of Environmental and Resource Economists (WCERE). The research outputs of this thesis will also be written up as empirical papers to be submitted to peer-reviewed scientific journals. The first empirical chapter will be submitted to the Journal of Environmental Management, the second empirical chapter to the Science of the Total Environment, and the third empirical chapter to the Environmental and Resource Economics.

Chapter 2. Literature Review

The present chapter reviews the relevant background information which is necessary to understand the economic perspective of value (section 2.1 and 2.2), as well as the particularities of the method used to measure the value of estuarine ES (section 2.3). Section 2.4 extends the literature review to describe previous applications of DCE to estuaries, along with the literature enquiring on ways to account for the diverse sources of environmental preference heterogeneity (section 2.5 and 2.6). Finally, section 2.7 identifies the gaps in the literature to be studied in this thesis.

2.1. The economic concept of value

Before understanding the approach we used to study society's environmental values, it is essential to define what is understood by 'value' and why something is 'valuable' from an economic perspective. The literature referring to the conceptualisation of 'value' recognise the existence of diverse approaches for studying the formation of value judgements (i.e. preferences). Even though all disciplines describe the valuation process differently, all definitions emphasise the role of 'values' as a conduct guiding principle (Rokeach, 1973; Schwartz, 1994).

Values could either be 'held' (by) or 'assigned' (to) ecosystems, environmental goods and services. *Assigned* values are developed through a continuous and complex cognitive valuation process in which internal and external factors are used to determine the scale of significance or desirability individuals ascribe to environmental commodities (Bingham et al., 1995; Papayannis and Pritchard, 2011). In contrast to *held* values, the *assigned* values are based upon relative valuation and rely on the formation of comparative judgements of value which are shaped by the context, as well as individuals' perceptions, preferences and beliefs (Brown, 1984). The extensive use of *assigned* values in valuation studies is justified as previous literature has suggested that they are comparatively better predictors of pro-environmental behaviour (PEB) when compared to the *held* values (Seymour et al., 2010; Tarrant and Cordell, 2002).

Society assigns values to all types of environmental commodities. Environmental goods and services often fall in the category of pure public goods, as they are commodities that can be consumed by various agents at the same time (non-rivalrous) and non-paying consumers cannot be excluded from their consumption (non-excludable). From an economic perspective, the existence of environmental public goods could lead to market failures which result in resource depletion, unless their users define sets of collective rules for managing their use (Ostrom, 1990). Nonetheless, developing management rules for public goods might be challenging (Battersby, 2017) as it is difficult to exclude from their consumption people who value them less (relatively to people who value them more).

Market prices reflect society's *assigned* values given to commodities (Brown, 1984), notwithstanding, many environmental goods and services have low or null prices in markets (but no absence of value). Under-pricing natural goods give wrong signals to producers and consumers regarding the scarcity of the good and the cost of the environmental damages associated with its consumption (or production). On the other hand, if ecological goods and services have no price and are non-marketed, both consumers and producers, face a marginal environmental cost of zero which indicates that reducing the stock of environmental goods by one unit have no cost to society. As a result of this market failure, the production and consumption of resource-depleting commodities are promoted, and more sustainable markets with resource-saving goods are hindered (Panayotou, 2013).

A major objective of environmental economic studies is to correct for the 'zero price' of the environment and to assess the economic value of environmental goods and services. Environmental economists study society's preferences through the analysis of individuals' *assigned* values and use non-market valuation techniques to understand ways in which society *"trade-off their environmental values in decision making"* (Seymour et al., 2010, p. 142).

In the environmental valuation framework, individual preferences and their *assigned* values are expressed via their WTP for ecological changes. Economic valuations of the environment are therefore developed with the purpose of generating a more efficient allocation and use of natural resources (Perman et al., 2003a), that for the case of environmental public goods is achieved when the social marginal benefits equal the social marginal costs. Valuation studies provide a further understanding of the social costs of environmental degradation and the opportunity cost of preserving the environment.

Environmental economics uses a ‘utilitarian’ or ‘instrumental’ approach to value, which considers that the value placed on any environmental good or service depends on their capacity to satisfy human needs and desire (Sinden and Worrell, 1979). Utilitarian analyses rely on Bentham’s theory of usefulness (Bentham, 1789) which not only limits the definition of this concept to its ‘practical’ dimension but also refers to the amount of pleasure or happiness caused by the consumption of a good. The ‘utilitarian’ perspective of value, despite being explicitly anthropocentric, is often misinterpreted as short-sighted as their critics fail to recall that an individual’s utility or happiness could depend on the utility of others (humans and non-humans beings from present and future generations) (see Turner, 2001).

Economic valuation studies using a ‘utilitarian’ approach do not obscure or replace ‘intrinsic’ values that society attach to nature for their complexity, rarity, spiritual significance, historical relevance or beauty. Instead, they serve to widen the understanding of the role that environment plays in society wellbeing (Daily et al., 2011). While valuing environmental goods, economists calculate an aggregated measure of values, which is known as the Total Economic Value (TEV).

The TEV framework considers six different categories of values (see figure 2-1 for details) that refer to ‘use’ and ‘non-use’ values. It is important to note that TEV includes benefits that do not have monetary contributions to society, reflected in all the non-marketed values of ecosystems and the option values (Kumar and Kumar, 2008). Moreover, this value framework considers an ‘existence’ value category which overlaps with all the culturally dependent values that are considered to be ‘intrinsic’ to all species and express ethical and aesthetical principles of protecting nature.

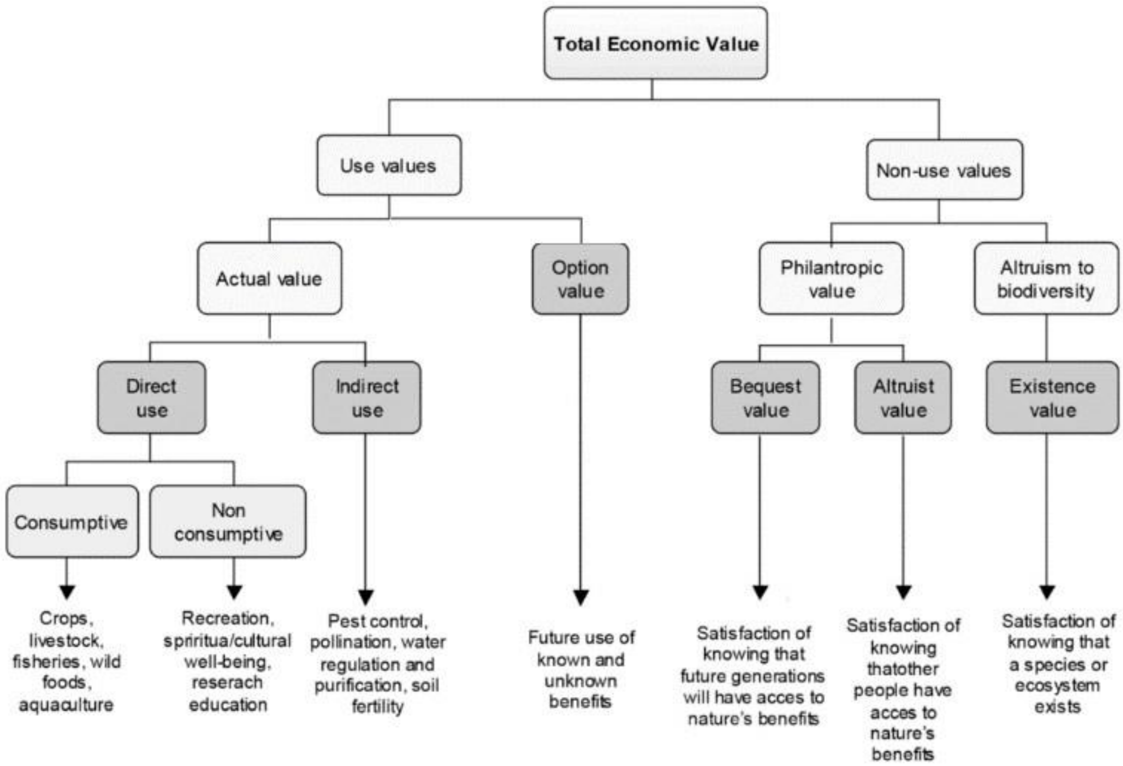

Figure 2-1 Types of values within the TEV framework. Source: (Pascual et al., 2010)

2.2. Economic environmental valuation

A central focus of attention in environmental economics relates to the correction of market failures that lead to negative externalities, such as environmental degradation and pollution. The primary purpose of using environmental valuation methods is to result in a more efficient allocation of resources. Valuation efforts intend to send correct signals to consumers and producers, and for doing so, they place a monetary value on environmental goods and services.

The measurement of environmental values has often used monetary units such as the WTP or willingness to donate (WTD), and willingness to accept (WTA). However, the valuation of the environment have also used non-monetary valuation units such as human health (e.g. QUALYs[2], DALYs[3], HYEs[4]); time (e.g. willingness to work[5]); energy (e.g. CO^2 tonnes, *emergy*[6]*)* and biophysical terms (e.g. m, kg, ha, person). While non-monetary units help to clarify the magnitude of some of the environmental assets, the use of units which are not "*readily intelligible and comparable to other benefits*" (DeFries et al., 2005, p. 54) impede their complete integration into the wellbeing-policy framework. Placing monetary values on environmental goods and services, on the other hand, offers a comparable, transparent and accountable information unit (Ozdemiroglu and Hails, 2016; Torres and Hanley, 2016).

Environmental valuation efforts assist decision makers with the integration of ecological goods into the policy framework, which already comprises the consideration of socioeconomic variables in the process of environmental planning (Frank et al., 2012). As Spangenberg and Settele (2010) acknowledged, valuation methods are also broadly used by environmental scientist to assist in transferring their message into the economic terminology understood by decision makers. Incorporating the monetary value of environmental benefits into a cost-benefit analysis (CBA) (Hanley and Barbier, 2009), for instance, could portray management policies safeguarding the natural capital as more desirable and provide a compelling argument for developing planning strategies which maintain or restore the environmental quality.

Environmental valuation methods have become a handy tool for decision making as they allow quantifying society's utility derived from the consumption of environmental goods and services. In other words, they permit measuring how much nature impact on society's well-being and happiness so that policies have better ways to prioritise, allocate and equally distribute benefits in society.

In Europe and the United States of America (USA), the welfare measures estimated through valuation studies have been used in CBA of environmental projects and policies,

[2] Quality-Adjusted Life Years (Hubbell, 2006)
[3] Disability-Adjusted Life Years (Cohen et al., 2017)
[4] Healthy-Years Equivalents (Hauber, 2009)
[5] Willingness to Work (WTW) is commonly measured in hours per month (Vásquez, 2014)
[6] Defined by Voora and Thrift (2010) as the energy that is used directly and indirectly to make a product or provide a service

in pricing policy, in the design of environmental taxes, or even, to develop participatory policy design exercises (Hockley, 2014; Pearce and Seccombe-Hett, 2000). Other examples of specific policy realms for which environmental valuations studies are applied worldwide include: i) addressing the shortcoming of environmental accounts for developing ‘green accounting’ practices in public and private sectors (Atkinson, 2010; UNEP-WCMC, 2011); ii) identifying, measuring and assessing value trade-offs and synergies resulting from land use changes (Kragt and Robertson, 2014; Wu et al., 2013); iii) detecting environmental value hot and coldspots (Johnston and Ramachandran, 2014; Meyerhoff, 2013) for setting priority ecosystems and/or regions; iv) improving the design and targeting of payments for ecosystem services (PES) schemes (De Groot et al., 2010; Karousakis, 2012); v) encouraging the use of access and benefit sharing (ABS) mechanisms between users and local providers managing an environmental good for its conservation (Prakash and Balakrishna, 2003); vi) re-evaluation existing economic and regulatory instruments such as tradable permits, compensation schemes[7], fines, subsidies and taxes to environmental commodities (Braat et al., 2008; UNEP-WCMC, 2011); finally, but not less importantly vii) facilitating the communication and understanding of society’s dependence on nature to both, policy makers and the general public.

Regardless of their particularities, all environmental valuation methods rely on an analysis of consumer preferences based upon welfare economics axioms (DeFries & Pagiola, 2005). This theoretical approach assumes that all individuals base their decision making on their preferences subject to income constraints and have the goal of utility maximisation. Valuation practitioners utilise value proxies to understand the welfare conditions generated by ES. Revealed preference (RP) methods observe consumers behaviour and employ market prices to infer the value of non-market environmental goods and services, whereas stated preference (SP) methods use hypothetical markets or scenarios to explore the elicited WTP to gain or avoid changes in their ecological conditions. In other words, they estimate the willingness to give up income or wealth to enjoy an increase in the provision of some ES or to avoid their deterioration.

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[7] Compensation schemes for ES losses (Sangkapitux and Neef, 2009), Biodiversity offsetting (Bull et al., 2013), Compensation payments for agri-environmental services in the European Union (EU) Common Agricultural Policy (Dedeurwaerdere et al., 2015)

The majority of the provisioning services are traded in and are hence valued by market forces. However, the valuation of cultural, regulating and supporting services require methods suited to the valuation of non-marketed goods. Explaining in detail the differences between all the existent valuation techniques considered within the RP and SP categories is out of the scope of our study. However, it is relevant to mention that in contrast to the former group of techniques, SP methods are capable of estimating both the use and non-use values of environmental goods, i.e. their TEV.

The two most popular SP methods are contingent valuation (CV) and DCE. Both of them are survey-based methods which estimate individuals' WTP for environmental changes. The main difference between them is that DCE uses an attribute-based approach to valuation, but CV does not. Therefore, the CV approach gathers information about respondent choice regarding a precise scenario, whereas the DCE approach is used to understand the respondents' preferences over the attributes describing that scenario.

Literature has discussed whether the accuracy of SP methods for estimating an individual's WTP depends on the reliability of the answers given to the presented hypothetical scenarios (List and Gallet, 2001; Little and Berrens, 2003; Murphy et al., 2005; Perman et al., 2003b). Empirical research has demonstrated that SP data is reliable (Bliem et al., 2012; Mørkbak and Olsen, 2015; Teisl et al., 1995). Nonetheless, in order to assess the 'hypothetical bias', SP data studies must include a set of core internal and external validity tests which should be interpreted within the proper theoretical and empirical context (Johnston et al., 2017). Following the best-practice guidance, for example, can help to increase the clarity and plausibility of the questionnaire (Bateman et al., 2002) and increase the 'content validity'. Furthermore, it is relevant to develop 'construct validity' tests where study results are compared with prior expectations. For instance, by doing the scope effect test and protest responses analysis (Bateman et al., 2004; Carson and Mitchell, 1993; Hanley et al., 2002a; Johnston et al., 2017; Meyerhoff and Liebe, 2008).

2.3. Discrete choice experiments

As explained previously, all environmental valuation methods have different capacities for calculating the welfare impact of environmental goods and services. A considerable effort in the environmental valuation literature has been focused on improving SP techniques to address methodological criticism levelled against them. The need for improving what was considered to be a well-established method for eliciting environmental preferences, i.e. the CV, became more relevant after the accumulation of evidence revealing significant problems related to it (Hausman, 2012; Kahneman and Knetsch, 1992).

Despite their popularity in marketing and transport research areas (Anderson et al., 1985; Louviere and Hensher, 1983; Louviere and Woodworth, 1983), survey-based methodologies utilising choice modelling (CM) approaches did not become an attractive alternative in the environmental research literature until the 1990s. That is, following the first application to natural resources by Adamowicz et al. (1994). Environmental valuation literature then started to argue for the 'adequacy' and 'superiority' of one variant of the CM approach, known as 'discrete choice experiments' to study society's preference-based values (see Hanley et al., 2001).

The DCE conceptual base relies on Lancaster's economic theory of value (Lancaster, 1966) and the random utility maximisation (RUM) theory (McFadden, 1973). The former theory explains an individual's utility derived from the consumption of a good as the composite of utilities associated with the characteristics or attributes of this good. The latter theory proposes the inclusion of random elements in the utility model (RUM models) to allow for the estimation of choice probabilities (McFadden, 2001).

In environmental applications of RUM models, an individual's utility for an alternative i depends on a deterministic component and a random component $U_i = V_i + \varepsilon_i = \beta\, x_i + \varepsilon_i$, where x_i is a vector of attributes describing the environmental management option and β is a vector of coefficients that explain the relative importance of the attributes to individuals. Individuals are assumed to select the management options which provides them with more utility. Therefore, the probability of choosing alternative i over j is $\mathrm{P}(i\ chosen) = P(V_i + \varepsilon_i > V_j + \varepsilon_j;\ \forall j \in C)$. The error term ε reflect researchers' inability to observe all the factors influencing respondents' choices for environmental management (McFadden, 1973), and randomness in choice on the part of respondents.

Further details of the different approaches used to estimate RUM models can be found in the sections 4.2, 5.2 and 6.2 of this thesis.

In a DCE, respondents are repeatedly asked to elicit their most preferred option when facing repetitive hypothetical choice scenarios of goods described in terms of their attributes and variations on their levels. The answers are used to estimate the model which predicts choice probabilities on the basis of an individual's willingness to trade between attributes. As the attributes are commonly used to define the characteristics of environmental goods or environmental policies, and one of the attributes usually represents the cost of this alternative, marginal values of a unitary change in any one of the attributes can be computed. The marginal WTP estimates are calculated with the ratio of the attribute coefficient to the estimate of the marginal utility of income (Train, 2009a).

The WTP estimate values, not only reflect individuals' potential monetary contribution but can also be interpreted as their 'behavioural intentions' (Bateman et al., 2003; Pouta and Rekola, 2001). Behaviour literature suggests that 'behavioural intentions' precede explicit behaviour and therefore are relevant to study for understanding and predicting social behaviour (Ajzen, 1985; Ajzen and Fishbein, 1980).

The application of DCE for assessing environmental preference-based values has practical and estimation advantages. Regarding the practical advantages, it is considered that the DCE technique is a realistic way of collecting preferences as it emulates real market situations where respondents are required to choose among alternative goods (Louviere et al., 2010). Second, the careful selection and design of attributes allow for increasing their credibility/viability and consequentially improve choice scenario realism (Hess and Rose, 2009). Third, DCE is considered to be a cost-efficient technique for measuring use along with non-use values (Birol and Koundouri, 2008; Hanley et al., 2001, 1998). The cost-efficiency of DCE is explained by their capacity to extract additional policy-relevant information such as the total and marginal values of several attributes (Hanley et al., 1998), as well as their capacity to derive multiple responses from each person surveyed (Hanley et al., 2001).

Regarding the estimation advantages, we can mention two relevant ones. First, the panel nature of the choice data permits to test the validity and consistency of respondent's answers throughout the repeated sampling of individuals (Boxall et al., 1996). Second, the DCE is considered to be a more robust method to avoid collinearity among attributes (Hanley et al., 1998), reduce *strategical bias* (Birol and Koundouri, 2008) and lessen ethical protests *"as the choice context can be less 'stark' than direct elicitation of willingness to pay"* (Hanley et al., 2001, p. 451).

On the other hand, the main criticism of using DCE for studying the decision heuristics relate to: i) the *hypothetical bias* caused by its 'fictitious' nature (Gómez-Baggethun and Ruiz-Pérez, 2011; Murphy et al., 2005), ii) the insensitivity of WTP estimate values to the scope or scale of attributes (Boyle et al., 1994; Kahneman and Knetsch, 1992), and iii) the effects of imposing a 'cognitive burden' to respondents (Swait and Adamowicz, 1996; Tversky and Shafir, 1992).

The problems mentioned above are not exclusive to this SP technique, and in fact, are partially mitigated because of the repetitive and additive nature of DCE (Foster and Mourato, 2003; Hanley et al., 2001). Furthermore, careful piloting of the design and modelling process help to alleviate these issues. Hence, the use of focus groups can help to define choice attributes, whereas *pilot* surveys are useful for pre-testing the survey and choice cards (Hoyos, 2010).

In their systematic review of the SP published literature, Mahieu et al. (2014) found that the probability for an article to use DCE during the years 2004-2013 is higher in comparison to the CV technique. Their analysis also revealed that this probability is relatively small in environmental studies when compared to studies from other research disciplines such as agriculture, health and transport. Their findings suggest that even though there is a growing popularity of this method, the use of DCE is still contested in the environmental literature. This could be partly explained by the ongoing debate in the literature around its capacity to account for the pluralism of values (Gómez-Baggethun and Ruiz-Pérez, 2011; Kumar and Kumar, 2008), as well as its ability to realistically represent and predict individuals' behavioural decision making process (Moshe Ben-Akiva et al., 1999; Spangenberg and Settele, 2010).

Simplifying the complexity of the decision heuristic is inevitable in any model, as they only serve as a *partial* representation of reality. The following section (2.4) in this chapter reviews the studies using the DCE to value estuarine ES, as well as the discrete choice literature exploring the sources of preference heterogeneity (section 2.5 and 2.6). In reviewing the valuation literature, we identified an emerging trend which claims for the use of more flexible modelling approaches which acknowledge the complexity of the human decision making process. Choice models have started to adapt to the study of intangible goods such as the estuarine ES, and have started to use behaviourally realistic structures for analysing preference-based *assigned* values. Thus, our research is in line with this novel body of literature as it contributes to revealing additional layers of preference heterogeneity.

2.4. Environmental valuation of estuarine ecosystems

The use of economic tools for assessing the estuarine ES has helped to recognise them as one of the most valuable habitats worldwide (Costanza et al., 1997). Additionally, it has helped to acknowledge that some of the benefits they provide are *"important economic and social imperatives"(Basset et al., 2013, p. 3)*. Some authors have suggested that using market prices as a proxy for the social worth of ES generate an inaccurate estimation of it, as it ignores the fact that prices for environmental goods depend on market conditions and regulatory policies (Barbier et al., 2011; Guimarães et al., 2011). As the majority of the estuarine benefits are non-marketed, the estuarine valuation literature has commonly used elicited values obtained with CV (Hazen and Sawyer P.C., 2008; Johnston et al., 2002a; Kroeger and Mcmurray, 2008) and DCE surveys (Birol and Cox, 2007; Hooper, 2013).

The DCE technique has been used previously in estuarine valuation studies worldwide. However, the studies using a choice modelling approach to value similar natural environments differ on the degree on which they account for the complexity of estuarine ecosystems structure and management.

The first group of studies base their level of analysis on valuing specific components of the estuarine ecosystems and aims to estimate society's WTP for preserving them. For example, the work of Hooper (2013) uses a DCE for estimating the WTP for reducing estuarine mudflat loss at The Taw Torridge estuary (North Devon, England). Another good example is the research of Boxall et al. (2012) which developed a DCE to estimate the WTP for implementing marine mammal recovery programs in the St Lawrence Estuary (Canada). In comparison, to the former study which is focused on valuing a structural component of estuarine ecosystems, the latter is interested in valuing biotic components of estuarine ecosystems.

The second group of researchers have focused their DCE designs on estimating the WTP for developing alternative estuarine management plans. A representative example of this type of research in the UK include Birol and Cox (2007), who valued wetland management alternatives in the Severn estuary aiming to result in larger wetland areas, otter hold creation, a high number of protected birds and higher levels of irrigation related employment. The work of Bhatia (2012) is another example in which SP techniques are used to value the societal benefits derived from implementing policies in estuarine environments, such as the development of four managed realignment sites on the Humber estuary, in the UK. This latter study uses a DCE to estimate the WTP values for the continued maintenance[8] of respondents' closest site, and access to it.

Finally, the third body of literature includes studies utilising the ES framework to value a single estuarine ecosystem. For instance, Vazquez and Iglesias (2015) analysed the WTP for tidal stream energy in in Ria de Ribadeo, an estuary in Spain. In addition to this, other researchers have focused on valuing cultural services derived from natural adventure tours in the Gironde estuary, France (Rambonilaza, 2011); recreational boating in the Kromme River Estuary, Eastern Cape (Lee et al., 2015); as well as recreational fishing in the Sundays River Estuary, South Africa (Lee et al., 2014).

Our review of the DCE literature applied to estuarine ecosystems showed that research has mainly focused on estimating the value of cultural services. Nonetheless, estuaries are considered to be one of the most productive ecosystems worldwide which provide at least 46 different ecosystem services (see list of estuarine ES in annex 1). Therefore, there is a need for developing research which identifies the values of other estuarine ES, as this

[8] As defined by Bhatia (2012), maintenance payments cover bank inspections and repairs; maintenance of all structures; clearance of any large debris and cutting the grass (i.e. keeping a generally pleasing aesthetic appearance). Additionally, it covers the development of regular ecological monitoring of the site, taking into account the vegetation, birds and fish assemblages.

research has proposed to. Moreover, it can be seen that much of the literature developed so far fails to account for estuarine ecosystems complexity. Environmental valuation studies, for example, tend to ignore the connectivity between different estuarine ES, or the interconnection of estuarine ecosystems with riverine and coastal habitats. Even though there are few environmental valuation studies which have employed DCE to estimate the value of bundles of ES in the UK riverine (Hanley et al., 2006; Stithou et al., 2011), marine (Börger et al., 2014; Jobstvogt et al., 2014a, 2014b) and coastal ecosystems (Acreman et al., 2011; Birol et al., 2009; Luisetti et al., 2011). To our knowledge, no DCE study has yet valued a bundle of ES in the context of estuarine ecosystems.

Finally, among all the estuarine valuation studies identified in the present literature review, only four studies have accounted for estuarine links with surrounding ecosystems by using a *catchment scale* analysis. That is the case of the work developed by Birol et al. (2009), Kragt and Bennett (2011a), Rolfe et al. (2004) and Stithou et al. (2011). The adoption of a *catchment scale* analysis is relevant when studying estuaries in Europe since the establishment of River Basin Management Plans is a requirement of the Water Framework Directive (200/60/EC), which goal is the protection, improvement and sustainable use of the water environment.

2.5. Environmental preference heterogeneity

Stated choices for ES management in DCE studies reflect individuals' environmental preferences. Discrete choice models represent a flexible analytical framework for valuing ES, but are also a suitable technique to understand the drivers of environmental preferences which intervene in the valuation process.

To quote from Ben-Akiva et al. (2002, p. 1267): *"[choice] models traditionally presented an individual's choice process as a 'black box', in which the inputs are the attributes of available alternatives and the individual's characteristics, and the output is the observed choice*". To date, numerous applications of the DCE method have been used to predict choice behaviour and generate environmental policy recommendations while neglecting the full spectrum of factors influencing the cognitive process of decision making occurring inside this *black box.*

The behavioural literature recognises two broad categories of factors influencing individuals' preferences, including i) the 'endogenous' and ii) the 'exogenous' factors, which will be referred in this text as 'internal' and 'external' factors.[9] In addition to this, we argue for the existence of a third category of factors that refer to iii) the spatiotemporal 'context' in which individuals develop the choice process, as both, the psycho-cognitive and socio-cultural factors explained above are dependent on their immediate context (Santos et al., 2011).

Internal factors (to individuals) mainly consist of psychological factors such as personal needs and motives (Börger and Hattam, 2017); personality types (Boyce et al., 2017) or self-identity (Van der Werff et al., 2013); attitudes (Mariel et al., 2015) and values (Maldonado-Hinarejos et al., 2014). Other internal aspects influencing preferences relate to individuals' cognitive capacity to represent and evaluate reality through perceptions (Bolduc and Alvarez-daziano, 2010).

On the other hand, external factors to individuals include socio-cultural factors (Kim et al., 2014a); sociodemographic characteristics such as education, religion (Hunter and Toney, 2005) and national identity (Kountouris and Remoundou, 2016); social status, networks (Kamargianni et al., 2014) and influences in the form of pressure and norms (Czajkowski et al., 2017a). As Von Auer (1998) points out, society's preferences are dynamic and could change with individuals' accumulation of knowledge and experiences (Ajzen, 2001).

Much of the refinement in the analysis of discrete choices has concentrated on the problem of how best to represent the heterogeneity of preferences across different individuals surveyed. The CM literature recommends different ways of exploring for the preference heterogeneity which depends on the variable of interest. If the aim is to reveal the effect of 'observable' or 'measurable' variables (e.g. socioeconomics) on taste variation, modellers use the observable variables in interaction with the attributes or the alternative specific constant (ASC) in the utility function (Train, 2009b). However, if the interest is to analyse the stochastic component of heterogeneity, it is then recommended

[9] This was done to avoid confusion with the two types of variables in macro-econometric models: often called endogenous and exogenous variables.

to use a logit model which include random coefficients in the utility function (Train, 1998). A combination of approaches can also be used.

The majority of environmental applications of DCE have used either or both of the previously mentioned modelling approaches, as they use information that is commonly collected in surveys and do not require additional modelling efforts. Nonetheless, the last decade has witnessed significant computational, informational and modelling developments, which allowed for the upsurge of innovative modelling frameworks for testing the effect of other types of factors, often referred in the literature as 'spatial' and 'latent' variables (Campbell et al., 2009; Czajkowski et al., 2016; Daly et al., 2012; Meyerhoff, 2013).

The choice modelling literature recently proposed the use of a two-stage approach (Abildtrup et al., 2013; Campbell et al., 2009) for studying preference heterogeneity (see chapter 5 for more details). In this approach, modellers specify the model in *WTP space* so that they directly obtain from the posterior analysis the individual-specific marginal values for each attribute as well as their distribution. Deriving individual-specific values is not only relevant for augmenting the level of detail of the analysis, but also allows for the use of individual-specific mean WTP estimates in further analysis. For instance, to assess for statistically different WTP estimates between sub-groups of respondents with similar characteristics. Furthermore, having data at the individual level facilitates its integration with other sources of data. One example of this is the use the analysis of WTP (geo-referenced) data in integration with relevant layers of 'spatial' variables or 'geographic' data through the use of geographical information systems (GIS) (see Meyerhoff, 2013).

The final innovative example in the CM literature relates to the use of 'latent' constructs within a hybrid framework of modelling. The Hybrid Choice Model (HCM) was developed by McFadden (1986) to correctly accommodate for the biological, psychological or sociological factors underlying the process of choice formation, often referred to as 'latent' variables. The hybrid structure of modelling is used so that latent variables that are 'intangible' or 'unmeasurable' concepts become 'observable' through its association with attitudinal indicators measured on a Likert scale. Hence, they can be inserted as interacting terms in the utility function to test for their effect on taste variation without taking the risk of developing biased estimates (Daly et al., 2012).

In the environmental valuation literature a vast number of SP studies have analysed preference heterogeneity analysis with respect to the internal ‘sociocultural’ dimension, which includes observed variables related to respondent’s socioeconomic characteristics (Andreopoulos et al., 2015a; Birol et al., 2006; Colombo et al., 2009, 2007; Hynes et al., 2008; Kragt and Bennett, 2011). However, a more recent and expanding trend in environmental valuation studies argues for using more complex and behaviourally realistic analysis which accounts for the presence of internal and ‘latent’ psycho-cognitive factors as attitudes (Bartczak et al., 2016; Boyce et al., 2017; Breffle et al., 2011; Hess and Beharry-Borg, 2012; Yoo and Ready, 2014). Finally, some authors accounted for the additional effect of ‘contextual’ factors such as social interactions (Kamargianni et al., 2014; Kim et al., 2014b, 2014a); as well as the local availability of substitutes, respondent’s relative location to the area and the geographical distribution of the good valued (Abildtrup et al., 2013; Brouwer et al., 2010; Garrod et al., 2012; Schaafsma et al., 2013).

Most of the discrete choice modelling studies have tended to focus on analysing one of the three factors influencing preferences that were previously explained. One attempt to include all three dimensions into the analysis was made by De Valck et al. (2012) who include what they defined as ‘individual-related’, ‘on-site’ and ‘off-site’ characteristics. Their work utilises interacted ‘observable’ and ‘latent’ variables with the covariates in the utility function of a standard choice model. However, their methodological approach has several drawbacks. First, it did not recognise the diverse nature of the variables and assumed that all of them are observable by the modeller and directly measurable. Second, the variables used as interacting terms might be correlated with the error term, leading to *endogeneity bias*. Finally, the addition of several interacting terms absorb specification errors and could generate inferior models (in terms of forecasts and welfare estimates) when compared to simpler models.

Further understanding of the factors taking part in the process preference and value formation is crucial to develop appropriate ways of modelling and measuring them. Developing adequate models to simultaneously integrate all the types of factors within the DCE framework could enhance the behavioural realism, but is out of the scope of this research objectives. However, this research intends to contribute with further insights on the topic, by developing independent models that aim to identify the significant factors influencing environmental choices and contributing to preference heterogeneity. The outputs of this research would be useful to understand some potential sources of preference heterogeneity; as in fact, integrating the whole spectrum of factors in DCE might be irrelevant if they are found to have no significant influence on individuals' environmental preferences.

2.6. Potential sources of preference heterogeneity

In the following sections (2.6.1 to 2.6.3), we review the literature focused on understanding the effect on taste variation of the three following factors: i) socioeconomic characteristics of individuals (i.e. visitor, resident), ii) latent attitudes (i.e. environmental consciousness), and iii) spatial context (i.e. local clustering of WTP estimates).

2.6.1. Socioeconomic characteristics

As explained previously, decision making is a complex process dependent on both, the individual's socio-cultural context and psycho-cognitive characteristics. It is well documented in the psychological literature that individual's characteristics such as age, levels of environmental concern (Honnold, 1984; Tarrant and Cordell, 2002; Van Liere and Dunlap, 1980; Wesley Schultz, 2001), gender (Brown and Reed, 2000; Hunter et al., 2004; Torgler et al., 2008; Wesley Schultz, 2001; Zelezny et al., 2000), education (Howell and Laska, 1992; Olofsson and Öhman, 2006), location and length of residency (Berenguer et al., 2005; Seymour et al., 2010), religion (Hunter and Toney, 2005; Sherkat and Ellison, 2007) and income (Diekmann and Franzen, 1999; Franzen and Meyer, 2010; Inglehart, 1995) influence the values that they *assign* to environmental goods.

Empirical psychological research has helped to put forward the recognition of socioeconomic factors in the value process formation, but have had difficulties in measuring the magnitude and direction of their effect. Therefore, discrete choice models which integrate socioeconomic variables into the analysis have become a suitable tool for filling this gap of knowledge.

The demographic and socioeconomic characteristics of individuals can have direct or indirect feedback on environmental preferences. In other words, socioeconomics might have a direct relationship to environmental preferences or be linked with variables that in turn impact the formation of environmental preferences (e.g. attitudes or beliefs). Obtaining information regarding respondent's sociodemographic (e.g. gender, age, the location of residence, number of visits to the area) and socioeconomic characteristics (e.g. educational attainment, employment status, household income) has been a common practice in DCE surveys.

This set of variables not only provide information regarding the demographic processes and portray respondent's state of wellbeing, but could also be used within the CM framework to i) test for their effect on systematic preference heterogeneity, or to ii) obtain the attribute coefficients for specific individual characteristics. Both types of analysis use socioeconomic variables as interacting terms in the utility function. The former test requires for the interaction with the ASC, whereas the latter demands the inclusion of socioeconomic variables in interaction with the choice attributes.

Choice experiments provide researchers with a useful modelling framework to test and measure the effect of socioeconomic factors on environmental preferences. The interaction effects have been analysed in discrete choice econometric models such as the multinomial logit model (MNL), random parameter logit model (RPL) and latent class model (LCM). The main drawback of MNL is that it does not allow for preference heterogeneity at the individual level as its utility parameters do not vary across individuals. The other two models do include random components in the model, RPL with a continuous distribution, and LCM with a discrete distribution. Even though previous research has suggested LCM dominance regarding welfare calculations (Birol et al., 2006), we opted for the use of RPL models in our analysis as our research objective is to reveal sources of preference heterogeneity within classes and not between classes. Moreover, RPL is more adequate to examine how individuals' preferences differ from the mean values of the utility parameters, i.e. from the average taste within the sample of respondents (see chapter 4).

Several empirical studies in environmental and resource economics have analysed socioeconomic variables in interaction with the ASC in the utility function (Börger et al., 2014; Botzen et al., 2012; Cerda, 2013; Garrod and Willis, 1998; Jobstvogt et al., 2014a; Shoyama et al., 2013; Stithou et al., 2011). There seems to be an agreement in their research findings which indicate that residents, females, younger, wealthier and more educated people have a stronger preference for environmental change or to avoid the *status quo*. These conclusions often extend to studies analysing management and conservation policies directed to estuaries (Kragt and Bennett, 2011) or to other ecosystems interacting with them, such as wetlands (Birol et al., 2006; Birol and Cox, 2007), rivers (Andreopoulos et al., 2015b) and coastal waters (Hynes et al., 2013b).

Among the previously mentioned variables and according to theory, income should be the most relevant variable in choice studies as it determines individual's budget and restricts the amount they would be willing to pay. Nonetheless, the empirical research has reported a positive income elasticity of WTP less than unity, which indicates that the proportion of WTP saved (from the total income) for funding policies improving ES decreases as income rise (Barbier et al., 2017; Jacobsen and Hanley, 2009).

The applications of DCE which explore the effect of socioeconomic variables on preference heterogeneity rarely account for the complexity of estuarine ecosystems and ignore that ES are commonly provided in bundles (Millenium Ecosystems Assessment, 2005). Moreover, they fail to recognise estuaries as transitional habitats and overlook its connections with riverine and coastal ecosystems. Differently from the previous literature, the work developed in chapter 4 increases the complexity of the analysis in several ways. First, we explore the effect of demographic and socioeconomic factors on the preferences for a 'bundle' of estuarine ES. Secondly, we use a scale of analysis that considers estuarine links with surrounding ecosystems, which is the *catchment scale*. Third, we develop a multiple case study analysis that increases the robustness of our research conclusions.

2.6.2. Spatial context

The provision of ES is closely related to spatial attributes of the natural environment (Bastian et al., 2012; Turner et al., 2013). The spatial variability of the ES supply is well-documented in the ecological and geographical literature and has been found to be dependent on the number, size, shape, connectivity or fragmentation of natural ecosystems (Haddad et al., 2015; Mitchell et al., 2015, 2013; Renó et al., 2016). Since natural habitats and natural resources are heterogeneously distributed in space, the provision of ES is also expected to vary across the territory.

The spatial attributes of the landscape impact the provision levels of the three ES of interest for this study. The spatial configuration of the land use in the catchment area, for instance, defines the capabilities of regulating river peak flows through surface run-off (Fohrer et al., 2001) and thus results in different levels of flood control. Similarly, the fragmentation of estuarine and coastal habitats by urbanisation and agriculture leads to considerable reductions in biodiversity (Fahrig, 2003; Thrush et al., 2008). This is due to changes in both, habitat structure and function; which restrict the dispersal of animals (Eggleston et al., 1999) and plants (Soomers et al., 2013); as well as diminish the habitat of native species (Marzluff, 2001). Finally, the frequency with which individuals participate in outdoor recreational activities is dependent on their access to the natural estuarine environment, as well as the distance from their home to the recreational sites (Koppen et al., 2014).

There seems to be an agreement in the literature about the relationship between landscape attributes and the variability of ES provision. Spatial preference heterogeneity might not only be related to the differences in ES supply across space, but could also depend on the levels by which these services are demanded or valued by society. For instance, the demand for natural flood control might be greater in regions that are highly populated, are lacking substitutes (e.g. dams), and where settlers have a greater appreciation of this service (risk propensity regions). To further characterise the geographic preference patterns and further understand the drivers of spatial preference heterogeneity, researchers have started to integrate GIS with environmental valuation studies.

The initial efforts to characterise the spatial heterogeneity of environmental preferences argued that global patterns (i.e. continuous patterns) were causing WTP variations. Global patterns refer to homogenous trends that are held within the limits of the area of analysis, and the valuation literature has explained them in two ways. First, in studies analysing a distance decay effect where the global pattern refers to a decline in WTP estimate values with an increase of the distance to the site of interest (Bateman et al., 2006; Cameron, 2006; Hanley et al., 2003; Pate and Loomis, 1997; Schaafsma et al., 2012; Schaafsma and

Brouwer, 2013; Sutherland and Walsh, 1985). Secondly, in studies which explain WTP differences in relation to respondent's relative location (inside/outside) to a defined geopolitical area (Aregay et al., 2016; Brouwer et al., 2010; Johnston and Duke, 2009; Kim et al., 2012; Morrison and Bennett, 2004). Empirical research investigating the existence of global patterns found mixed evidence with results varying with the type of good and when comparing use vs non-use values (Hanley et al., 2003; Martín-López et al., 2007). Furthermore, these relationships are difficult to conceptualise and test for intangible goods the distribution of which cannot be constrained to specific coordinates, as it is the case of various ES.

The strong assumption of global patterns of environmental preference has recently been challenged in the SP literature. Some authors suggest that environmental preferences are more likely to be explained by local association forces that generate 'patchy' patterns (Johnston et al., 2015, 2011; Johnston and Ramachandran, 2014; Meyerhoff, 2013). Local spatial patterns are thus described by the presence of 'non-continuous' or 'clustered' patterns of WTP variations across the area of analysis.

There are several reasons why we would expect that environmental preferences present local (i.e. discontinuous) rather than global (i.e. continuous) spatial patterns. First, the spatial context is individual-specific which means that people might develop a positive emotional connection or 'place attachment' to their local and familiar context (Manzo, 2005, 2003). Second, individual WTP reflects the scarcity (or abundance) of the ES in their immediate environment (Bockstael, 1996; Johnston et al., 2002b), as well as the local availability of and accessibility to substitutes (De Valck et al., 2017; Jørgensen et al., 2013). In this sense, the spatial pattern of preference heterogeneity might be partly explained by the underlying local distribution of the supply of ES. Thirdly, the cultural and socioeconomic characteristics which could impact society's WTP for environmental improvements also vary at the neighbourhood level. Therefore WTP patterns might follow the local spatial configuration of wealth, education levels, employment rates, cultural identity or environmental consciousness, which are reflected in the demand side of ES. Finally, the existence of local WTP clusters might be a result of preference clusters that arise from residential sorting (Baerenklau et al., 2010; Timmins and Murdock, 2007), which suggest that individuals chose their residence location in according to their environmental preferences and the costs of relocation.

The spatial dimension of environmental preferences has been explored in various stated and revealed preference studies which use CV (Jørgensen et al., 2013; Kim et al., 2015; Pate and Loomis, 1997; Söderberg and Barton, 2014; Sutherland and Walsh, 1985); hedonic pricing (Anselin and Le Gallo, 2006; Cameron, 2006; Cavailhès et al., 2009; Paterson and Boyle, 2002); as well as the travel cost technique (Bateman et al., 1996).

Within the choice modelling framework of analysis, the spatial approach has been developed in four ways: i) using spatially explicit choice attributes, ii) including spatial covariates in the choice model, iii) applying geographically weighted choice modelling, and iv) developing a second-stage spatial analysis. The first approach considers the spatial dimension at the experimental design phase and utilises spatially explicit choice attributes and choice cards (Brouwer et al., 2010; Horne et al., 2005; Johnston et al., 2002b; Schaafsma et al., 2012). The main disadvantage of this approach is that it can result in very complex designs that impose an additional cognitive burden on the respondent.

The second alternative is to include interactive terms with spatial covariates describing respondent's location (Bergmann et al., 2008); or their distance to the service provision source and substitutes (Meyerhoff, 2013; Schaafsma et al., 2013). However, this approach might result in overparameterised models with multicollinearity problems.

A third approach, recently proposed by Budziński et al. (2017) applies geographically weighted models in analysing discrete response variables so that a nonlinear relationship with respect to spatial dimensions is recognised while modelling choices. Nonetheless, for estimation purposes, this approach assumes global spatial autocorrelation of WTP estimate values, which has often been found to be low or not statistically significant for the case of environmental goods and services (Johnston et al., 2015, 2011; Johnston and Ramachandran, 2014; Meyerhoff, 2013).

The fourth alternative is the one selected for this research and makes use of a two-stage analysis to examine the effect of spatial variables on individual-specific WTP (see chapter 5 for more details). Different techniques have been used for exploring spatial determinants of WTP in the two-stage approach of analysis. For instance, Vollmer et al. (2016) used a non-parametric locally weighted scatterplot smoothing technique to contrast WTP estimates with the distance variable. Other authors have estimated panel random effects regressions with distance (Campbell et al., 2008; Johnston and Ramachandran, 2014; Yao et al., 2014) and accessibility (Abildtrup et al., 2013) as spatial covariates. The studies of Budziński et al. (2017) and Czajkowski et al. (2016) argued for the use of a GIS-based spatially lagged regression to accommodate for spatial dependence. The two-stage approach has not only been applied for analysis purposes, but also have assisted in the visualisation of the geographical distribution of WTP estimates. For instance, Johnston et al. (2015) used inverse distance weighted interpolation to visualise the raw spatial patterns of the sampled points, whereas Campbell et al. (2008) and Czajkowski et al. (2016) used regression kriging to extrapolate values and create a smooth surface of predicted values based on spatial dependence in a non-sample.

Finally, some authors have combined the strategies described previously to accommodate spatial effects. A prominent example of this is the work of Meyerhoff et al. (2014) who developed choice cards with maps indicating the stretches of the rivers targeted for water quality and subsequently include distance and location variables as interacted covariates in the choice model.

Table 2-1 summarises some of the characteristics of the studies applying a spatial approach to analyse environmental preferences. This table shows that research on spatial preference heterogeneity has tended to focus on environmental goods, rather than ES. Moreover, studies have not yet established whether there are similarities between the local spatial patterns of different environmental goods and services. With this in mind, chapter 5 presents the analysis developed for contrasting the spatial patterns of local clusters of WTP estimates among estuarine ES and across catchment areas.

Analysis testing for local spatial autocorrelation has been widely developed in other disciplines, and to a lesser extent in the environmental valuation literature. In the context of the latter body of literature, the analysis of local clustering permits to measure different concepts of spatial association between WTP estimates, such as the spatial clustering of *similar* values and the spatial clustering of *dissimilar* values (Anselin, 1995).

Table 2-1 Summary of environmental choice experiment studies using the spatial approach of analysis

Author	Technique	Good or service valued	WTP space	Two-stage analysis	Description	Location
Abildtrup et al. (2013)	RPL/EC	Forest recreation	No	Yes	Study using estimates in a second-stage analysis to estimate the potential spatial determinants of the preferences for forest recreation, as well as including spatial attributes into their DCE.	France, Lorraine
Budziński et al. (2017)	GWDC & RPL	Forest attributes	Yes	Yes	Study comparing geographically weighted discrete choice models with the two-stage analysis. Additionally, it uses the individual-specific WTP estimates as dependent variables in a GIS-based spatial regression.	Poland
Campbell et al. (2008)	RPL	Rural landscape improvements	No	No	Study exploring spatial autocorrelation of individual-specific WTP.	Republic of Ireland
Campbell et al. (2009)	RPL	Farm landscape improvements	No	Yes	Study using kriging methods to extend WTP estimates across the whole of the Republic of Ireland.	Republic of Ireland
Czajkowski et al. (2016)	RPL & LCM	Forest attributes	Yes	Yes	Study using spatial econometric methods and high-resolution GIS data related to forest characteristics are used to explain individual-specific WTP estimates.	Poland
De Valck et al. (2017)	LCM	Nature restoration	No	No	Study using a respondent-centric approach to control for substitute sites and assessing for each respondent-specific spatial context by computing densities of nature substitute sites within various ranges from each respondent's home.	Belgium, Flanders
Johnston et al. (2014)	RPL	Migratory fish passage restoration	No	No	Study developing methods to explore local patchiness and hotspots in SP welfare estimates using local indicators of spatial association.	US, Rhode Island
Johnston et al. (2015)	RPL	Threatened or endangered marine species	No	No	Study demonstrating an expanded suite of methods that may be used to identify and characterise discrete, multiscale heterogeneity in stated preference WTP, including the coordinated use of spatial interpolation (IDW interpolation) with multiscale analysis of hotspots and coldspots using local indicators of spatial association.	United States
Meyerhoff et al. (2013)	RPL	Wind power generation	No	No	Study using GIS to investigate whether spatial heterogeneity affects choices regarding the future shape of wind power generation on a regional level.	Germany
Meyerhoff et al. (2014)	RPL	Water quality improvements	No	No	Study using DCE designed in a spatially explicit manner by using river stretches as choice attributes.	Germany, Berlin
Sagebiel et al. (2017)	RPL	Local afforestation	No	No	Study examining the spatial distribution of WTP for changes in forest cover and proposing a novel method to derive spatially explicit WTP	Germany

Author	Technique	Good or service valued	WTP space	Two-stage analysis	Description	Location
					which uses of a function for predicting predict the mean WTP at the county level, which depends on the status quo and other spatial variables.	
Vollmer et al. (2016)	RPL	Rehabilitation and conservation of river catchment	No	No	Study applying the spatial approach in developing country	Indonesia, Jakarta
Yao et al. (2014)	RPL/EC	Habitat improvement for enhancing biodiversity	No	No	Study adding spatial covariates in second stage analysis of determinants of WTP.	New Zealand

RPL Random Parameter Logit (RPL), Random Parameter Logit plus Error Component Logit (RPL/ECL), Geographically Weighted Discrete Choice (GWDC), Latent Class Model (LCM).

2.6.3. Environmental attitudes

Following Betsch and Haberstroh (2012, p. 102) definition, attitudes are *"the feelings and evaluations associated with a representation of an object"*. Environmental attitudes can thus be understood as the aggregations of evaluations or beliefs which individuals place towards the natural environment.

Attitudes have a role in the decision making process as they consist of stored evaluations or feelings influencing how individuals assess the available options and the overall choice situation. In integration with other constructs[10], attitudes are a guiding principle to behaviour when they are used to value the behavioural consequences and the expectations towards actions (Betsch and Haberstroh, 2012). For instance, society's attitudes towards choices of environmental management could be based on the perceived uncertainty of the choice outcome (Lundhede et al., 2015). Additionally, they could serve as a guide for filtering information regarding the choice alternatives which lead to the immediate avoidance, refutation, or rejection of all the facts that contradict pre-existing attitudes (Frey, 1986; Houston and Fazio, 1989; Lord et al., 1979). Ultimately, by serving as a mechanism to generate evaluations about the choice situation they determine choice behaviour, at least to a certain extent.

The tendencies to engage in evaluative processes vary among individuals (Jarvis and Petty, 1996). Hence some people are more likely to hold well-defined attitudes towards their natural environment than others. Personal attitudes are also shaped by the accumulated knowledge and experiences that vary with individuals' socioeconomic characteristics (Ajzen, 2001). Since society holds heterogeneous attitudes towards the environment, it is expected that these differences would also be reflected in the decision making process. This research explores this idea further, and for doing so, it tests the effect of latent attitudes on preferences for policies managing estuarine ES (see chapter 6).

The initial efforts to account for attitudes in discrete choice models originate in the transport literature in the late 1970s (Koppelman and Pas, 1980; Prashker, 1979). By then

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[10] Such as social norms, personal goals and routines.

models inserted perception indicators (Green, 1984) or latent attributes (Keane, 1997) directly into the utility function. These approaches were criticised because of their forecasting limitations (Kløjgaard and Hess, 2014). Moreover, this approach was found to generate inconsistent (Ashok et al., 2002) and biased estimations (Hess and Beharry-Borg, 2012) as self-reported attitudinal indicators have a measurement error and might be correlated with unobserved factors in the error term. To deal with this, a new approach was proposed by McFadden (1986) and extended by Ben-Akiva et al. (2002, 1999), which integrated discrete choice models and latent variable models in a hybrid modelling framework.

The development of two precursor models was essential to construct the structure of the model named by Bolduc et al. (2005) as the Integrated Choice and Latent Variable (ICLV) model and referred to here as the HCM. Firstly, the choice models considering Structural Equation Models (SEM) which reduces model dimensionality and explain the latent variable in terms of exogenous observable variables, such as the individual's socioeconomic characteristics. Secondly, the development of measurement equations embedded in Multiple Indicator Multiple Cause (MIMIC) models which relate the latent variable with psychometric measurement indicators to become observable (Bollen, 1989).

Although the past few decades have experienced an increase of applications of the so-called HCM in several disciplines (e.g. transport, agriculture, medicine), its use in environmental valuation studies is still limited. The popularity of this model type in environmental valuation studies might be hampered by the increase in the modelling and computational effort they represent. Table 2-2 lists the studies using HCMs in the environmental valuation literature. This table reveals that the majority of the studies modelled choices while accounting for taste heterogeneity and used either Hybrid Mixed Logit (HMXL) or Hybrid Latent Class (HLC). It can also be seen that the minority of studies (38%) considered the ordered nature of the attitudinal questions, as suggested by Daly et al. (2012).

The applications of the HCM framework in the environmental valuation literature have mainly focused on three topics. Empirical studies are mainly testing for the impact of pro-environmental attitudes on choosing greener transport and energy alternatives (Daziano and Bolduc, 2011; Hess et al., 2013; Hess and Beharry-Borg, 2012; Maldonado-Hinarejos et al., 2014; Mariel et al., 2015). The second group explores the effect of attitudes and perceptions on preferences for environmental improvements (Boyce et al., 2017; Czajkowski et al., 2017a; Faccioli et al., 2018; Hess and Beharry-Borg, 2012; Santos et al., 2011). Finally, the rest of the studies analyse how prior constructs of uncertainty and risk affect preferences for biodiversity conservation programs (Bartczak et al., 2016; Lundhede et al., 2015).

To the present, environmental valuation studies using HCMs have defined the environmental changes as an improvement in the adoption of an environmental policy, or enhancements on the ecological quality of an ecosystem. However, there has been little discussion on the effect of latent variables on preferences for the provision of one or several ES. Our research fills this gap in the literature and presents an analysis of individual's WTP for improvements in flood control, biodiversity and recreation in estuarine ecosystems; while at the same time investigating the impact of the latent variable called 'environmental consciousness'.

Similarly to Jiménez Sánchez and Lafuente (2010) and Zelezny and Schultz (2000), we defined environmental consciousness as the psychological factors influencing individual's tendency to engage in PEB. The behavioural literature has found a direct relationship between environmental attitudes and *environmentally conscious* behaviour (Alwitt and Berger, 1993; Berger and Corbin, 1992; Corraliza and Berenguer, 2000; Hines et al., 1987). Moreover, marketing research studies have suggested that more *environmentally conscious* individuals are more likely to develop ecologically conscious consumer behaviour (ECCB) (Antil, 1984; Roberts and Bacon, 1997; Shetzer et al., 1991). Therefore this study aims to test if this finding extends to situations where individuals are required to elicit their choices for a range of environmental management policies delivering improvements in the provision of estuarine ES.

HCMs integrate latent constructs and discrete choice models to account for the impact of cognitive and psychological variables on the decision making process (see chapter 6 for more details). They represent an alternative modelling framework to understand the underlying factors that affect choices at a cognitive level. Several studies have claimed that using this model result in gains of explanatory power and efficiency improvement that comes from having new information and an additional structural relationship in the HCM framework (Bolduc and Alvarez-daziano, 2010; Kim et al., 2014a; Kløjgaard and Hess, 2014; Maldonado-Hinarejos et al., 2014; Vij and Walker, 2016). However, recently some authors started to question their capacity to generate substantial statistical benefits (Chorus and Kroesen, 2014; Daly et al., 2012; Kløjgaard and Hess, 2014; Vij and Walker, 2015). In their systematic analysis of the statistical properties of the HCM, Vij and Walker (2016) concluded that using hybrid models lead to appreciable improvements in fit and predictive power when the observable explanatory variables of the structural equations (e.g. socioeconomic variables) are poor predictors of the latent variables, and therefore modellers are using 'truly latent' concepts. Hence the prediction gains are obtained from using measurement models with attitudinal information that offer additional insights about the choice outcomes.

Although estimation improvements are valuable in the CM literature, the most crucial advantage of using HCM refers to its capacity to generate additional policy-relevant information that could not be obtained from the reduced form model (Vij and Walker, 2016). For instance, we can understand the magnitude and the direction in which the latent attitude impact respondent's choices and their answers to the attitudinal questions. Furthermore, it is also possible to explore how sociodemographic factors relate to this latent attitude and to identify what factors should be targeted by policies if the aim is to change environmental attitudes, and behaviour.

Table 2-2 Summary of environmental valuation studies using hybrid choice models

Author	Technique	Good or service valued	Description	Latent variable	Indicators	Location
Bartczak et al. (2016)	HLC	Conservation and survival of endangered lynx populations	Studied the impact of individual risk preferences on conservation policies of threatened species	Latent risk preferences	Binary logit	Poland
Boyce et al. (2017)	HMXL	Environmental quality improvement	Studied the effects of personality on individual economic choices over public environmental goods	Extraversion; agreeableness; conscientiousness; neuroticism; openness to experiences	Ordered logit	Latvia and Estonia
Czajkowski et al. (2017)	HMNL and HMXL	Waste management	Study linking statements over attitudes to recycling to choices	Social Norms, Morals and Self-interest	Ordered probit	Poland
Daziano et al. (2011)	HMNL and HMXL	Low-emission vehicles	Bayesian approach to simultaneous estimation of hybrid choice models	Environmental consciousness	Binary logit	Canada
Facciolli et al.	HMXL	Peatland restoration management options	Explored the simultaneous role of environmental attitudes and place identity perceptions on individual's WTP	Environmental attitudes; place attachment	Ordered probit	Scotland
Hess et al. (2012)	HMNL	Water quality improvements	Use a latent attitudinal variable to explain both, the responses from the stated choice exercise as well as answers to respondent attitudes	Pro-intervention attitude	Continuous (zero centred)	Tobago,
Hess et al. (2013)	HLC	Reductions in GHG emissions in rail travel	Studied how the allocation of a given respondent to either choice class is a function of underlying attitudes	Pro-environmental attitude	Ordered and binary logit	UK
Hoyos et al. (2015)	HLC	Land-use policies in Natura 2000 Network site	Used a psychometric scale, the awareness of consequence scale to understand stated choices and class allocation	Beliefs supporting environmental action; beliefs supporting environmental inaction	Ordered logit	Basque Country, Spain
Lundhede et al.(2015)	HMNL	Conservation policies	Studied how prior perceptions of uncertainty affect preference for conservation	Uncertainty assessment	Continuous (zero centred)	Denmark!

Author	Technique	Good or service valued	Description	Latent variable	Indicators	Location
Maldonado-Hinajeros et al. (2014)	HMXL	Cycling demand	Analyse the impacts of attitudes and perceptions on cycling demand	Pro-bike (environmental and sustainability aspects); image; context; stress	Not specified	London, UK
Mariel et al. (2015)	HLC	Wind turbines	Investigate public sensitivities towards wind turbines by incorporating individuals' attitude in the latent class structure	Attitude towards wind power generation	Ordered and binary logit	Germany
Santos et al. (2011)	HMXL	Water and sanitation	Model explicitly the cognitive process that influences water and sanitation technology adoption choices	Attitude towards sanitation	Not specified	Salvador, Brazil

Hybrid Multinomial Logit (HMNL), Hybrid Mixed Logit (HMXL), and Hybrid Latent Class (HLC).

2.7. Conclusion

The review of the relevant literature has allowed us to identify a need for using experimental designs and modelling approaches which account for both the complexity of estuarine ecosystems, as well as the complexity of individual's behavioural decision making process while analysing their preferences for ES.

We identified three knowledge gaps that need to be filled. First, there is a need to explore the effect of demographic and socioeconomic factors on preferences for ES, in the context of estuarine ecosystems. Second, there is a need to understand the effect of the latent attitudes on estuarine ES preferences, such as the degree of environmental consciousness. Finally, there is a need to understand further how the local context can influence spatial patterns of WTP for estuarine ES.

The use of discrete choice modelling in integration with socioeconomic, attitudinal and spatial variables is a suitable analysis tool for filling these gap of knowledge. Thus, this research applies three different analytical methods, or CM approaches, which are presented in chapters 4, 5 and 6. These analyses were developed to evaluate the effect of socioeconomic characteristics, latent environmental consciousness and the local spatial context (respectively) on respondent's WTP for restoring the provision levels of three policy-relevant ES: flood control, biodiversity and recreational services. The following chapter (3) details the design of the DCE used in this research.

Part III. Ecosystem services preference heterogeneity

Chapter 3. Material and methods

This study developed a DCE for exploring respondents' preferences for estuarine ES management. Chapter 2 has provided a historical overview of the method, a brief explanation, and exposed some of the reasons for which it is considered to be a superior technique for valuing environmental goods and services. The description of the choice model particularities will be described in each data chapter (4, 5 and 6), starting from the simpler to the more advanced models, respectively. Instead of presenting the theory related to this method, this chapter outlines the process for adapting the DCE technique to our study objectives. The method adaptation is particularly relevant as we are using a multi-case study and require a more complex analysis framework that permits the inclusion of observable socioeconomic characteristics, latent attitudes and spatial variables.

The rest of this chapter is organised as follows. Sections 3.1 and 3.2 detail the choice experiment design. Afterwards, section 3.3 describes the survey instrument used, as well as the data collection and pre-processing procedure. Finally, section 3.4 and 3.5 present some survey results and summarise the characteristics of the study sample, respectively.

3.1. Choice experiment design

Choice experiments are a well-established technique for measuring use and non-use values in the environmental valuation literature (Birol and Koundouri, 2008; Hanley et al., 2001, 1998; Mahieu et al., 2014). Researchers develop DCE to further understand the effects of the attribute levels on individuals' stated preferences (Mangham et al., 2009). The development of a choice experiment comprises several stages which are summarised in figure 3-1. This chapter aims to describe and justify the decisions made in each of these design stages.

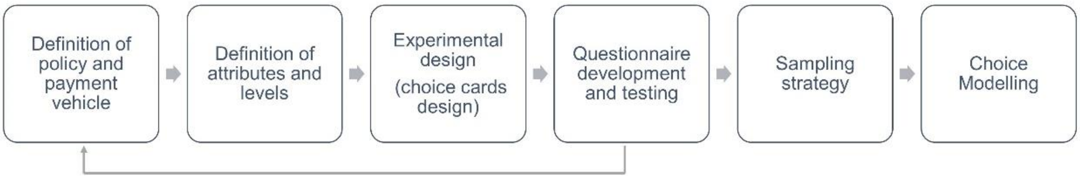


Figure 3-1 Methodology summary

The grey line connecting step four and one indicates that the information obtained from pre-testing the questionnaire could be used to re-define the design of DCE.

3.1.1. Case studies

This research studies three estuaries that are relevant in Scotland because of their socioeconomic and ecological characteristics. The definition of catchment area used in this work is consistent with the Scottish Environment Protection Agency (SEPA) classification, as our analysis considers what they define as the ‘Advisory Group Area’ for the Clyde, Forth, and Tay estuary (see figure 3-2).

The Tay catchment area is the most extensive study area, with approximately 9126 km^2. It includes the river Tay along with its tributaries (e.g. river Garry, Tummel, Lyon, Braan, Dochart, Erich, Isla, and Almond), as well as the catchments of river Dighty, Cowie, Bervie, river North Esk and South Esk; river Earn and river Eden together with Eden estuary. River Tay is considered to be the longest river in Scotland, covering an area of 5000 km^2 and 190km in length. Perth and Dundee are the most populated cities, followed by smaller settlements such as Arbroath, St Andrews, Forfar, Montrose, Carnoustie, Stonehaven, and Cupar (National Records of Scotland, 2014). Pitlochry is also an important town as it is a popular tourist destination. The Tay catchment area is mainly rural and comprises significant environmental assets distributed along 457,474,900 ha of forest woodlands and inside 13 national parks, eight national natural reserves, 28 special area of conservation, 18 special protected areas and six national scenic areas.

The Forth catchment area covers 4658 km^2 along the central belt and to the eastern coast of Scotland. It contains all the area draining into river Forth, as well as river Leven, Devon, Allan Water, Teith, Forth, Carron, Avon, Almond, Water of Leith, Tyne and Esk. The highest populated settlements inside the Forth catchment area are Edinburgh, Falkirk, Dunfermline, Livingston, Cumbernauld, Kirkcaldy, Stirling, Glenrothes, and Dalkeith (National Records of Scotland, 2014). This region comprises diverse land uses including managed forest and farmland, as well as natural heritage areas of national and international importance. It includes 395,107,415 ha of forest woodlands, three world heritage sites, one national park, five national natural reserves, 12 special area of conservation, ten special protected areas and two national scenic areas.

Finally, the Clyde catchment area has an extension of 7445 km^2. It contains the catchment areas of river Clyde, Kelvin, Leven, White and Black Cart Waters, Ayr, Irvine, Doon, water of Girvan and river Stinchar. This area encompasses contrasting landscapes that range from the largest populated settlement in Scotland, the city of Glasgow, to scenic natural areas such as Loch Lomond and the Trossachs National Park. Contained in the Clyde catchment area, there are 779,677,326 ha of forest woodlands, five national parks, 15 national natural reserves, 19 special area of conservation, nine special protected areas and two national scenic areas. Other largely populated settlements in the area are Motherwell and Belshill, Coatbridge and Airdrie, Hamilton, East Kilbride, Greenock, and Ayr.

The natural environment of the three selected Scottish estuaries is impacted by a range of economic activities and the regular dredging necessary for navigation purposes. The Clyde and the Forth estuaries have higher levels of pollution that result from historical discharges from, for example, pulp and paper mills, iron and steel manufacturing, and chemical plants (Scotland’s Environment, 2011). The Clyde estuary is considered the most degraded of all three as it presents instream and riparian habitats severely damaged and has a high profile of non-native species (Clyde River Foundation, 2009). The Tay estuary, on the other hand, still holds a rich natural heritage with semi-natural ecosystems and numerous resources providing habitat for rare wildlife, but is facing an increase in pressures from the agriculture, forestry and hydropower generation (Tayside Biodiversity Partnership, n.d.).

Current anthropogenic pressures present in the Clyde, Forth and Tay catchment areas are leading to a worsening of the environmental quality of the three estuaries. The absence of an adequate environmental policy that addresses this problem has the potential to alter estuarine capacity to provide all types of ES, including the ones of interest for this research that is flood control, biodiversity and recreation.

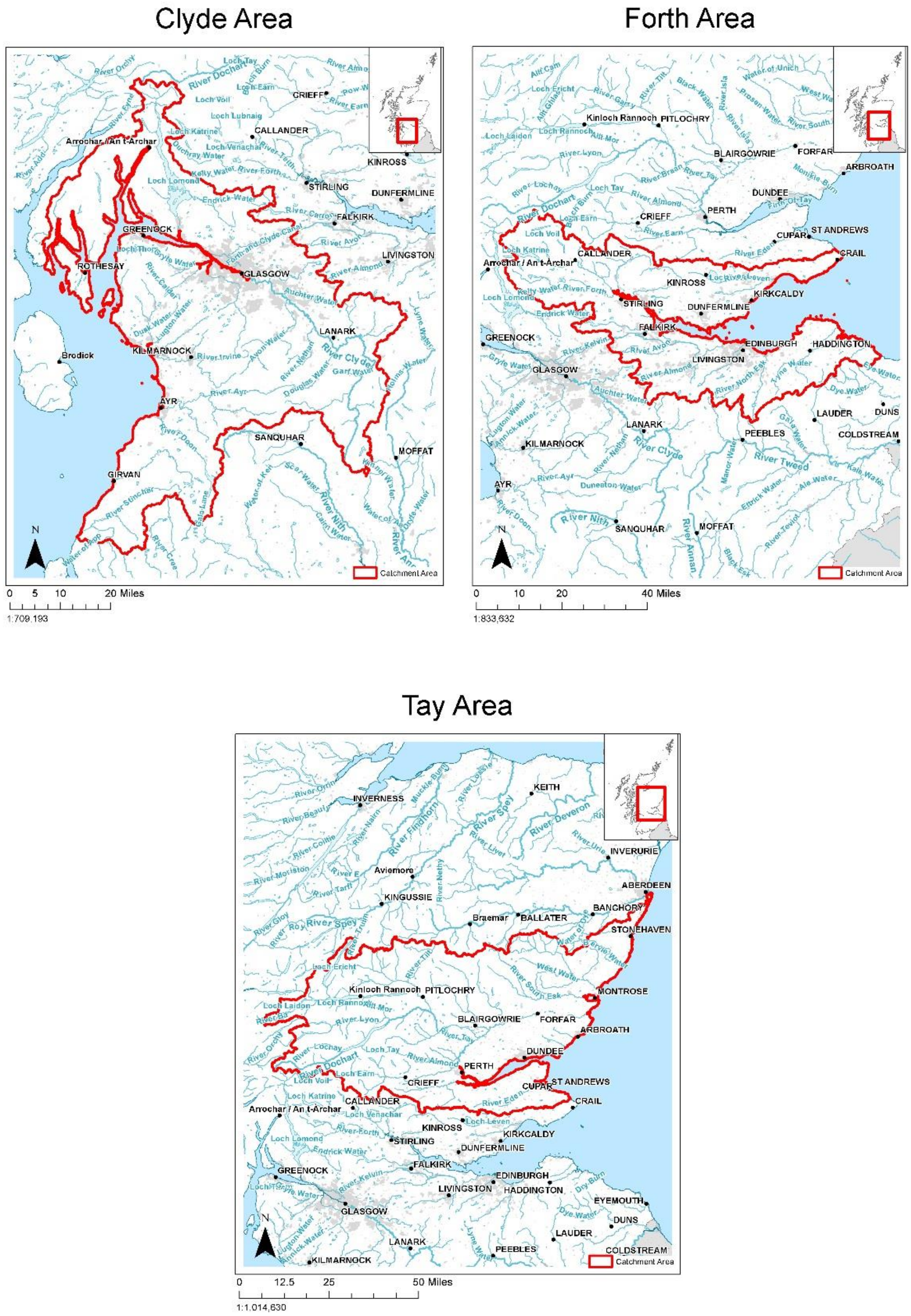


Figure 3-2 Catchment areas studied

3.1.2. Policy and payment vehicle definition

This study uses a DCE for estimating individuals' annual WTP for improvements in flood control, recreation and biodiversity delivered with the restoration of the catchment areas of the Clyde, Forth or Tay estuary, in Scotland. In a DCE respondents need to have incentives to state their true WTP. Therefore, respondents need to be presented with a viable and credible project; moreover, they need to understand *what* is going to happen, *how it is going to affect* the environment and *how it is going to be funded*.

This study uses a *catchment scale* analysis which acknowledges the linkages of estuaries with riverine and coastal ecosystems. The catchment management unit has been previously considered for legislation and policy tools which aim to protect the water environment quality at the European Union level, such as the Water Framework Directive (200/60/EC). It has also been applied at a national level, with the River Basin Management Plans For The Scotland River Basin District (Natural Scotland, 2015) and The Scottish Biodiversity Strategy (Scottish Executive, 2004). The geographic limits of the catchment areas analysed in this study are consistent with the also called 'Area Advisory Groups', which are the geographic areas used by SEPA to deliver basin management plans.

A *restoration project* was chosen as the hypothetical policy as it consists of an integrative strategy capable of delivering improvements on the provision level of all three ES, simultaneously. All the measures suggested as part of this restoration project were explained in a visual and textual format in the survey. For this, we used two illustrations representing the scenario before (figure 3-3) and after (figure 3-4) the restoration project (in each catchment separately). The illustrations included some callouts with text detailing the corresponding restoration measures and their consequences regarding the ES provision (for details see full questionnaire in annex 8).

The *final* objective of the restoration project is to improve the physical condition of the natural environment and to reverse historical damages at the catchment level. The restoration policy chosen is flexible enough to adapt to the multiple case studies context, as it is vital that respondents believe the viability of developing the policy on the Clyde, Forth and Tay context. Estuaries are complex and dynamic ecosystems influenced by the ocean, riverine and land management, therefore require management plans that account for their interactions and feedbacks with other ecosystems (Jacobs et al., 2014).

The use of the DCE technique is not straightforward and requires further understanding of the ecological functioning of the ecosystem being valued. Obtaining this information is not trivial when the ecosystem in consideration is as complex and dynamic as estuarine ecosystems are. Therefore, we proposed a management option which is aligned with the Supplementary Plan for River Basin Management Plans by Natural Scotland (Natural Scotland, 2013), the Integrated Coastal Zone Management (Scottish Coastal Forum, 2004), and Scotland's National Marine Plan (The Scottish Government, 2015). Finally, the restoration project attains improvements in environmental quality without compromising the socioeconomic development, as it “sets out a [technically] feasible and proportionately [expensive] approach to prioritising improvements in the water environment actions” (Natural Scotland, 2013, p. 6).

Respondents were told that the project would be financed with increased local taxes. We specified the UK council tax as the *payment vehicle* and suggested a one-time increase in the annual local tax (council tax), lasting for ten years. This *payment vehicle* has been previously used to value environmental policies in the regional and national UK context (Birol and Cox, 2007; Garrod and Willis, 1998; Hanley et al., 2006; Luisetti et al., 2008) and unlike voluntary donations, it does not encourage ‘free-riding’ behaviour (Whitehead, 2006).

Figure 3-3 Catchment area before the restoration project

Figure 3-4 Catchment area after the restoration project

3.1.3. Attributes and levels definition

A vital step in designing a DCE is the selection of adequate *attributes and levels* used to describe the ranges of the environmental changes which result from implementing the environmental policy. Both elements play a key role in different parts of our methodology, as they are part of the individual's utility model, but are also used during experimental design. Additionally, they define the units of measure and could influence the estimates as they determine ways in which respondents perceive the trade-off between money and the environmental good (Torres et al., 2011).

The application of economic valuation studies to the environment places a more significant challenge when compared to other research areas because they often deal with unfamiliar goods and use complex environmental policies. The present study is not an exception to this as it values three ES. We carefully designed the attributes and levels so that they represent not only viable scenarios of ES provision, but also portray meaningful scenarios. In this sense, attributes were designed to facilitate the understanding of a difficult concept, such as ES, by the general public.

Environmental choice analysts are required to effectively communicate the policy choices in terms of their characteristics, and yet consider appropriate numbers of attributes which offset the respondent's cognitive load related to task complexity (Hoyos, 2010). Focus groups were not developed due to financial restrictions. Instead, we consulted experts on the topic and developed an extensive literature review so that we include relevant attributes and use appropriate ways to describe their levels. Previous environmental valuation studies have used more than one attribute to describe ES such as flood control (Botzen et al., 2012; Koetse and Brouwer, 2016; Makriyannis et al., 2018; Reynaud and Nguyen, 2013; Ryffel et al., 2014; Zhai et al., 2007); biodiversity (Beukering et al., 2008; Christie et al., 2006; Dissanayake and Ando, 2014; Do and Bennett, 2007; Horne et al., 2005; Kragt, 2009; Meyerhoff et al., 2009; Nordén et al., 2015); and recreation (Christie et al., 2007; Hanley et al., 2002b; Juutinen et al., 2011; Lee et al., 2013; Naidoo and Adamowicz, 2005; Rulleau et al., 2010; Yan et al., 2008).

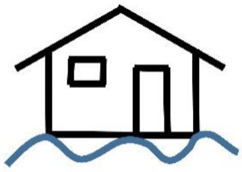
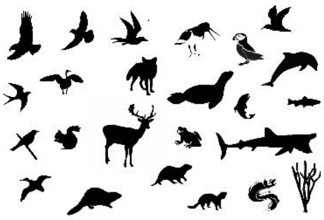
Using multiple attributes per ES can be useful when the aim is to value particular components of an environmental policy that delivers ES improvements. However, we decided on rejecting this approach and instead describe each estuarine ES with only one attribute for the following reasons. First, describing three ES with several attributes each would result in a large number of attributes. It has been suggested that as the number of attributes rises it is more difficult to obtain a good experimental design (Mansfield and Pattanayak, 2007) and respondent's choice consistency is affected (Deshazo and G., 2002). The former is explained by potential correlation issues (especially between biodiversity and recreation) that could arise when disaggregating each ES into several attributes. Whereas the latter is related to the imposition of a higher cognitive burden to respondents since the choice cards have more attributes and present higher task complexity (Hoyos, 2010; J Meyerhoff et al., 2014; Swait and Adamowicz, 2001). Another relevant reason for not using several attributes per ES type is that our research is focused on understanding the relative values of the three selected ES (i.e. ES attributes' weights), rather than valuing specific components of the ES restoration policy (i.e. policy attributes' weights).

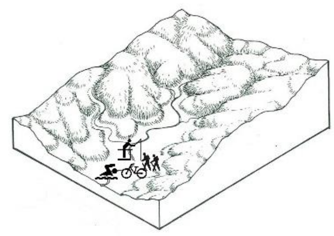
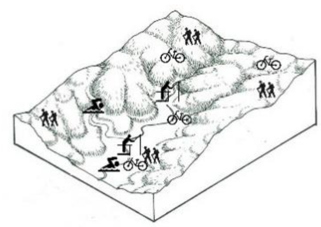
The use of site-specific attributes and levels was not employed in our study, because we develop a further comparison of the environmental preferences among the Clyde, Forth and Tay case studies. Instead, we opted for the use of four qualitative and generic attributes, including i) flood control (reduction in flood risk), ii) biodiversity, iii) recreation, as well as a final monetary attribute describing the iv) annual cost for developing the restoration project. The monetary attribute is necessary to produce welfare analysis and to estimate attribute's values in pound sterling (GBP). Respondents' WTP reflects the assigned values to attributes for a unitary change in them.

Choice attributes can be described with qualitative or quantitative levels. However, the development of multiple case studies imposed some limitations on the design of our attributes. Using the same quantitative levels (e.g. numbers, percentages or ratios) for describing the changes in estuarine ES provision at the three catchment areas might, for example, result in meaningless and un-realistic policy scenarios. This is because all regions have different current baseline scenarios. Qualitative levels, on the other hand, are useful for conceptualising water management options (Birol et al., 2009b, 2006; Hanley et al., 2006) as they can be conveniently abstract so that respondents translate what does the defined change means with respect to the current site-specific baseline level.

To ease respondents' cognition, we used three broad qualitative levels that define the possible changes in the magnitude of estuarine ES provision, but we carefully described to respondents what we meant for each of them (see table 3-1). All attributes and levels were presented with their corresponding descriptions and images to convey the information in various formats. The three non-monetary attributes used are the i) decrease, ii) slight improvement, and iii) large improvement on ES provision. The monetary attribute has six levels ranging from 5 to 100 and describes the annual fixed increase on the council tax (lasting ten years) that respondents would incur if they choose any of the alternatives developing the restoration project. The maximum payment level (£100) only represents 0.36% of the average annual earnings in Scotland (Office for National Statistics, 2015).

Table 3-1 Attributes, levels and illustrations used in the DCE

Attributes	Levels	Definition	Illustration
Flood control	Increase in flood risk	The frequency of flood events keeps increasing in the area (more events each decade and more chances each year). Flood defences fail because straightening rivers and the absence of vegetation keeps a high-speed flow of water. The failure to provide a free space between the river and human activities (buffers) also mean that the extent of residential and agricultural damages keeps increasing in time.	
	Slight reduction in flood risk	Flooding occurs every fifty years in the area. Already installed flood defences are useful because the restoration of the curvy shape of rivers helps to reduce speed flow. The extent of residential and agricultural damages is reduced significantly as buffers are created in some areas.	
	Large reduction in flood risk	Flooding occurs every hundred years in the area. No need for new flood defences as the restoration of the river shape and vegetation (upstream, in river plain and riverside) helped to lower speed flow. Residential and agricultural damages have almost completely been avoided with the creation of buffers.	
Biodiversity	Decrease in biodiversity	The chances of observing any type of wildlife (fish, birds, butterflies, mammals or reptiles) are reduced in the area because habitat degradation continues. Endangered species disappear.	
	Slight increase in biodiversity	Improvement in chances of observing birds, butterflies and few mammals happen when restoring ecosystems with native vegetation in the area. An increase in the observable number of endangered species happens inside protected areas.	 1
	Large increase in biodiversity	Improvement in chances of observing fish, birds, butterflies, mammals and reptiles happen when restoring ecosystems and eliminating structures that act as barriers to wildlife movement. An increase in the observable number of endangered species happens inside and outside protected areas.	

Attributes	Levels	Definition	Illustration
Recreation	Decrease in recreation	The quality of outdoor recreation decreases. Degradation of nature has led to non-scenic areas. Insufficient and not well-maintained infrastructure hinders access to the riverside and shoreline. Wildlife watching is possible everywhere but no walking, cycling, recreational fishing, swimming and other water sports.	
	Slight increase in recreation	Restoration and greening policies have increased the scenic quality and access. A path network with multi-purpose trails and resting places has been developed in few isolated areas, improving its quality of outdoor recreation. Wildlife watching, walking, cycling, recreational fishing, swimming and other water sports is possible ONLY in those areas.	
	Large increase in recreation	Restoration and greening policies have increased the scenic quality and access. A path network connects woodland, cities and coast with multipurpose trails and resting places. Wildlife watching, walking, cycling, recreational fishing, swimming and other water sports can be developed all around the area. The quality of outdoor recreation increases everywhere.	
Cost of the policy	Six annual payment levels	One time increase on the lasting for ten years.	**£5, £10, £20, £50, £75, £100**

3.2. Experimental design

The *experimental design* is done to obtain different attribute combinations which reflect the potentially expected policy outcomes. Statistically efficient designs are fractional factorial designs used for minimising the standard errors of parameter estimates (Bliemer and Rose, 2011). Choice experiment literature has considered different types of efficient designs which differ on the optimisation criteria applied.

Table 3-2 presents a summary of some of the most relevant criteria for developing efficient designs. DCE literature suggests that D-efficient designs outperform other design criteria (Carlsson and Martinsson, 2003; Hess et al., 2008; Scarpa and Rose, 2008). Among the disadvantages of using D-efficient designs are the requirements of *a priori* knowledge related to parameter estimates priors and the utility function (Rose et al., 2009). However, D-efficient designs are becomingly increasingly popular as the available statistical software ease its generation, they require small samples to achieve the desired standard errors, they allow for design constraints (Bliemer and Rose, 2010a), and lastly, they are invariant to the scale of the parameters (Street et al., 2005).

Table 3-2 Efficient design criteria

Criteria	Description	Optimize given
D-efficiency	Maximise the determinant of the information matrix	Sample size
C-efficiency	Minimise the variance of the marginal WTP estimate	NA
S-efficiency	Maximisation of the minimum t-ratio over all parameter estimates	Sample size
B-efficiency	Avoid choice sets containing alternatives that may be strongly dominated	Utility balance
A-efficiency	Maximise utility balance	Sample size

Evidence suggests that efficient designs help to increase data quality, estimation reliability and sampling cost-effectiveness (Scarpa and Rose, 2008). However, considering an adequate number of attributes and levels is crucial, since respondent's response efficiency could worsen in cases where the dimensions of the experimental design are not simplified (Johnson et al., 2013; Jordan J. Louviere et al., 2008). As for the case of D-efficient designs, the evidence favours the use of *end-point designs* which use two levels with wide range (Louviere et al., 2000; Rose and Bliemer, 2013). Moreover, Rose and Bliemer (2013) recommend using more than three attributes so that the design is less likely to have dominant alternatives (i.e. those with a high probability of being chosen) which do not provide much information about the choices and therefore increase D-errors.

We followed the suggestions of Rose et al. (2009) for generating a D-efficient serial design, but we adapted their work to a multi-case study which accounts for the site-specific environmental preferences. Figure 3-5 summarises the experimental design process used in this study.

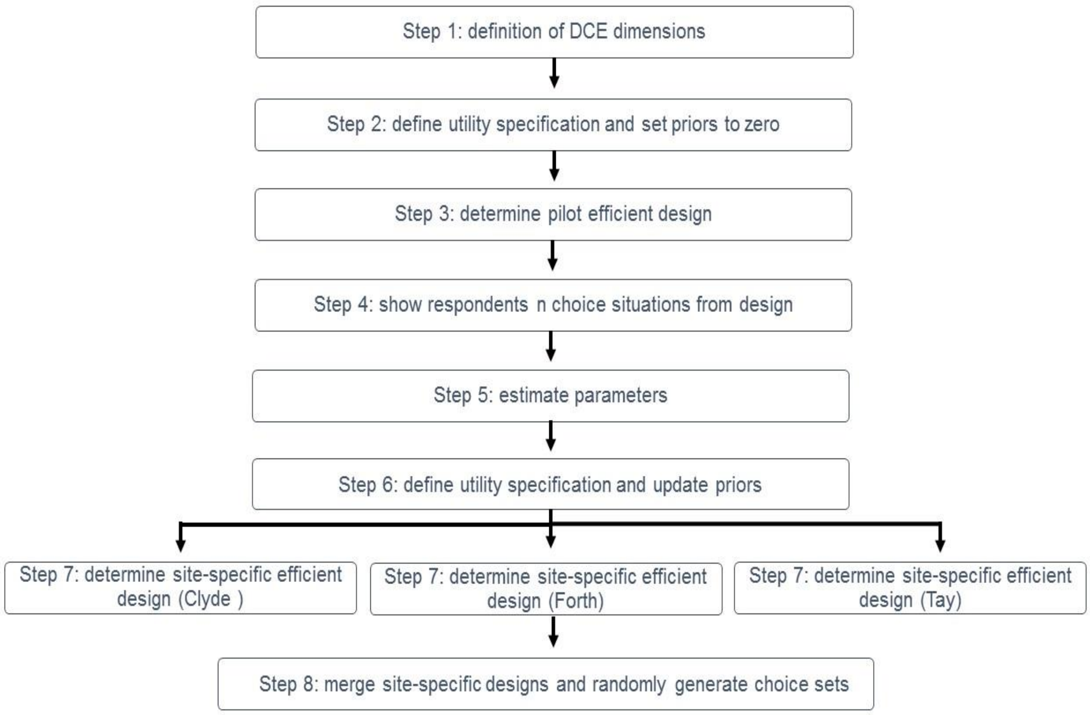


Figure 3-5 Experimental design generation process

Step 1 consists of the definition of the dimensions of the DCE. For this, we used the command *%mktRuns (3**3 6**1)* in the SAS software Version 9.4 (SAS Institute Inc, 2014) to find sample sizes in which a perfect balance (equal sample sizes per combination) and orthogonality can occur (correlations between effects is zero). This command finds the number of runs required by an orthogonal array for the specified number of attributes and levels and displays alternative arrays from a vast catalogue of designs. Details of the algorithms used by this command can be found in Kuhfeld and Tobias (2005).

The *%mktRuns* output suggested that eighteen was a reasonable number of choice tasks to include in our experimental design (see annex 2). The eighteen choice cards were grouped in three blocks (or treatments) so that each respondent would have a reasonable number of choice tasks to answer (six choice tasks per person).

Once the dimension of the DCE was set to four attributes[11] and six choice cards with three management alternatives (see figure 3-6), we proceeded to the *Step 2* and defined the utility specification for the *pilot* experimental design (annex 3). Bliemer and Rose (2010b) have shown that experimental designs using a MNL model offer similar efficiency when compared to those using RPL models. In the *pilot* study, we were interested in estimating the main effects of a simple MNL model. Thus the utility function does not contain interactions. All attributes were generic to all choice alternatives and were given no priors (zero as priors). Finally, the ES attributes were inserted with design coding (e.g. 0, 1 and 2) and the cost attribute with their actual numeric levels (e.g. 5, 10, 20, 50, 75 and 100).

In Step 3 several pilot experimental designs were generated with Ngene Econometric software Version 1.1.0 (ChoiceMetrics, 2012) and were evaluated using the D-error measure. The alternative management options were restricted so that they presented improvements of at least one estuarine ES and were not identical to the *NO policy scenario*, but with a positive cost. The experimental design that minimised the D-error to 0.02 (see annex 4) was tested in a *pilot* study of 58 individuals during *Step 4*. Approximately a third of individuals answered the Clyde, Forth, and Tay *pilot* questionnaires, respectively. The *pilot* study permitted us to pre-test the questionnaire, the choice context and the experimental design, as well as rectify respondent's understanding of background information, choice context and the additional questions.

After collecting the *pilot* choice data, we moved forward to *Step 5* and modelled the choices of each case study separately to obtain three sets of site-specific attribute coefficients. The resulting parameter estimates are influenced by the statistical efficiency of the experimental design, as well as the individual's response efficiency (Johnson et al.,

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[11] Three ES attributes with 3 levels each, plus a monetary attribute with six levels.

2013). The specification of the better-fitted *pilot* models include the ES attributes in a dummy coded format, the cost in a numerical format, and the ASC as a dummy variable (zero if respondents did not choose the *status quo* option and one if they did).

The *Step 6* starts with the second loop of the previously explained steps and is therefore equivalent to *Step 2*. It requires updating the utility function of the experimental design by adding the estimated *pilot* coefficients as priors. In order to avoid the loss of information of the site-specific environmental preferences throughout the process leading to the *final* experimental design, we generated one experimental design per case study (see annex 5). The utility functions of these site-specific D-experimental designs estimated the main effects of a simple MNL and included priors for the dummy coded ES attributes, the scaled numerical cost (divided by 10) and the ASC coefficient.

In *Step 7* (equivalent to *Step 3*) we used the D-efficiency criteria to select three *final* site-specific experimental designs of eighteen rows each[12] (see annex 6). We obtained three sets of 18 cards (per case study), which gives a total of 54 unlabelled choice cards. Unlabelled choice cards lead to smaller experimental designs (have less possible combinations) and are considered more robust as their alternatives are less correlated with the attributes (Bekker-Grob et al., 2010).

The experimental design process of this study aims to account for the site-specific environmental preferences. Thus, instead of blocking the experimental designs per case study, in *Step 8* we merged all the site-specific designs into a pooled-design of 54 choice cards. These were randomly grouped in unique choice sets (or blocks) of 6 choice cards, which were afterwards inserted in the DCE survey. Randomly choosing rows from an experimental design matrix provides even greater variation than using a limited number of blocks, and permits the composition of choice sets with choice cards that account for the site-specific preferences (Czajkowski, 2016).

The choice cards presented people with three management alternatives and asked them to choose their most preferred option. The first option always represents the *status quo* alternative. The *status quo* represents the absence of a restoration policy and means no

[12] D-error of 0.36 for Clyde, 0.31 for Forth and 0.33 for Tay.

additional financial cost to respondents, but would result in the decline of ES over time (UK National Ecosystem Assessment, 2011). The second and third options of estuarine ES management represent alternatives leading to improvements in the provision levels of at least one ES and thus are associated with a positive cost. All alternatives used generic titles to be identified (e.g. Option 1, 2 or 3). An example of an unlabelled choice card is depicted in figure 3-6.

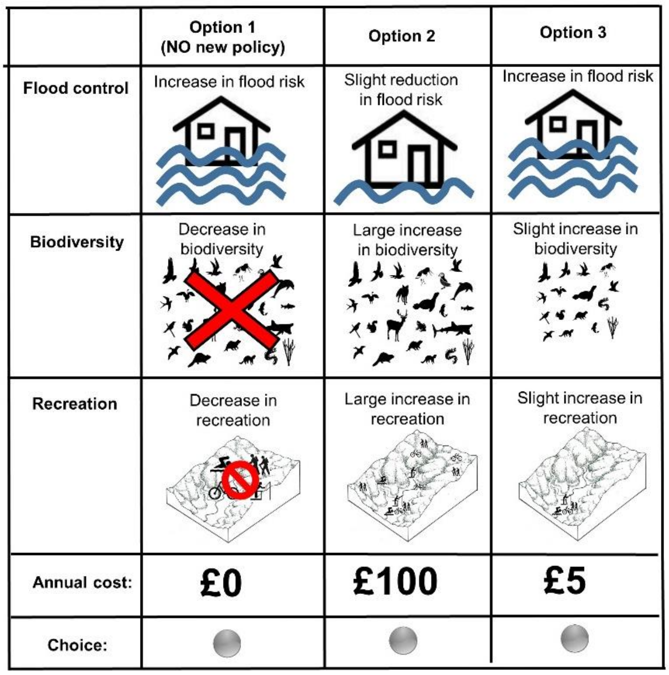


Figure 3-6 Choice card example

3.3. Questionnaire development, sampling design and data collection

The *final* experimental design was used in the survey to collect preferences towards ES. We used a self-administered web-based survey designed with Sawtooth Software's CBC/Web system (Sawtooth Software, 2008) which was distributed in September 2016 by the market research company Toluna (response rate 72.47%). The University Teaching and Research Ethics Committee (UTREC) approved this study from an ethical point of view (see annex 7).

Internet-based surveys using online panels are becoming increasingly widespread in non-market valuation studies as they are advantageous in their marginal costs, speed and response rates (Cobanoglu et al., 2001; Olsen, 2009). In addition to this, online surveys tend to be more flexible to customisation and are advantageous for the analysis process, as they automatically input the data and consequently reduce coding or inputting mistakes (Hess and Rose, 2009). Finally, as Hess and Rose (2009) rightly point out, online surveys present one choice card per screen and thus make it impossible for respondents to develop cross-scenario comparisons of the alternatives.

There are also advantages of using self-administered online surveys, including the participant's capacity to control their response pace (Champ and Welsh, 2006). Furthermore, self-administered surveys reduce the *social desirability bias* that occurs when respondents distort their preferences to create a favourable impression on the surveyor (Karina Gallardo and Wang, 2013; Leggett et al., 2003).

Three *final* versions of the survey were used; which differ in the case study they are referring to (Clyde, Forth or Tay catchment area) and the choice tasks contained. The sampling was not restricted to individuals living within the catchment area due to limitations on the size of the online research panel associated with each of them. More importantly, because individuals might care about the levels of ES provision within the Clyde, Forth and Tay area, even if they live somewhere else. Therefore, all versions were randomly allocated to people living in Scotland and being at least 18 years old.

We used quota sampling for monitoring the number of respondents answering each version of the survey, and to obtain similar samples between them. Hence, all cases have the similar representation in the *final* sample. Individuals could be answering the questionnaire of an area they reside in or not, and as a result, our survey collected both use and non-use values. Excluding non-use values may lead to underestimations of the values *assigned* to the ES, as the empirical evidence indicate that the passive use value represents a significant component of the TEV of environmental goods (Loomis, 2006).

Figure 3-7 depicts the structure of the DCE survey, and the full detailed questionnaire can be found in annex 8. The *final* questionnaire consists of four sections. The first section presents a brief explanation of relevant concepts, the environmental issue and the restoration project proposed to address it. The second section assesses environmental perceptions and previous knowledge on the topic. The third section includes the DCE together with a set of follow-up questions that serve to differentiate between genuine and protest bid individuals. Finally, we include a section with debriefing and consistency enquiries, followed by questions recovering respondents' socioeconomic characteristics.

The median time respondents used for answering the two surveys are similar, since they took 13.10 and 14.23 minutes for answering the *pilot* and *final* survey, respectively. The mean, on the other hand, is more affected by outliers with is 20.23 minutes for the *pilot* survey and 41.20 minutes for the *final* survey.

Mathews et al. (2006) argue that the use of background information becomes more relevant in surveys valuing less familiar goods or services, such as ES. In the opening section of the survey, we explained what an estuary is and the geographical limits of the catchment areas containing them. Additionally, we exemplified the ways in which the current management is affecting the benefits society obtain from them.

As in any other type of survey, it was vital to find a balance between the quality and quantity of the background information. The survey was tested several times prior to the online launching to avoid 'wording issues' and reduce measurement error (Dillman et al., 2014). Feedback was used to adapt the background information to the general level of familiarity with the topic and to adapt the language to a more accessible format. The *final* questionnaire uses un-ambiguous, neutral and factually based information to avoid biasing the results (Rea and Parker, 2005; Rossi et al., 1990).

In DCE surveys it is recommended to include questions that retrieve respondent's experience with the good that is asked to be valued (Krupnick and Adamowicz, 2006). Respondent's experience with goods could be influenced by personal attitudes, as well as the degree on which they benefit from a specific ES. This study used questions that identify respondents' type of user (active or passive). Additionally, it included attitudinal questions which could serve as covariates, class segmentation variables, 'psychometric' measures (Ben-akiva et al., 2002), or as variables to generate categories of preferences heterogeneity. Rating scale questions were also included to identify respondent interest or perception about the status of the ES. Finally, the survey used questions evaluating respondents' understanding of the most relevant concepts in the survey, which are flood control, biodiversity and recreation.

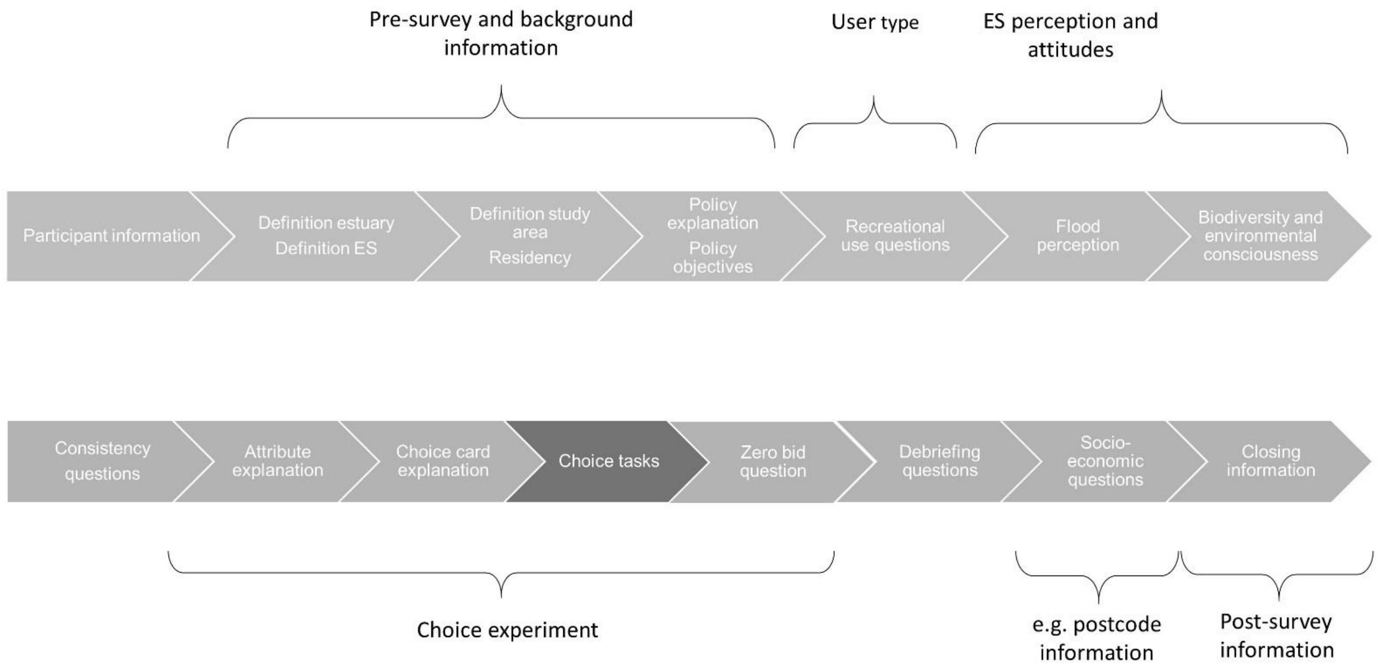


Figure 3-7 Survey structure

Studies using SP surveys have been criticised because of the presence of *hypothetical bias,* which refers to the existence of inconsistencies between respondents' hypothetical and real behaviour. Literature has emphasised on the relevance of using 'cheap talk' scripts to reduce this bias in choice experiments (Bosworth and Taylor, 2012; Carlsson et al., 2005; Cummings and Taylor, 1999, 1998). The cheap talk scrips commonly state to participants that there is a propensity of respondents to overstate their WTP and remind them about their budget limitations so that they select the options they can afford. The work of Tonsor and Shupp (2011) revealed that these scripts also have a significant effect on the WTP estimates for the case of DCE conducted via online surveys. Therefore, the third section of our questionnaire included a cheap talk script prior to the presentation of the six choice cards (see annex 8).

Following the standard practice, we included a post-DCE section with debriefing questions which retrieve "essential information needed to interpret responses and results, delete observations, and shore up the credibility of the survey" (Krupnick and Adamowicz, 2006, p. 53). In our survey, debriefing questions were used to i) obtain further information about the choice process (e.g. choice strategy, acceptance of time scale), ii) explore the acceptance and comprehension of the information in the text, iii) obtain opinions about the survey bias, and to iv) explore sample validity. The consistency questions, on the other hand, permit us to i) explore the quality of the answers and to ii) validate choice responses. The final section of the survey collects sociodemographic and personal information that was included as covariates in the modelling stage (see chapters 4, 5 and 6). This section was included after the DCE to avoid influencing choice responses (Hess and Rose, 2009).

3.4. Survey results

Table 3-3 shows the numbers of respondents choosing the *status quo* alternative in all their choices and the statements used to identify 'protest bid' individuals, for whom the *"zero amount was not considered to be a true reflection of the respondent's value" (Hoevenagel and van der Linden, 1993, p. 232)*. The share of respondents always choosing the *status quo* does not differ significantly between the three case studies. The percentage is 1.98% for the Clyde, 1.97% for the Forth and 1.48% for the Tay.

After removing the ‘protest bid’ individuals (1.80%) and deleting respondents without postcode information (1.32%), we obtained the choices of a *final* representative sample of 589 individuals. Each of them answering six choice cards, meaning that we obtained 3,534 choice observations (approximately a third for each study case).

As Kuhfuss et al. (2015) indicate, low numbers of protest responses are desirable (see table 3-3) as it suggests that respondents consider the hypothetical scenarios to be credible and agree on the use of taxes as the payment mechanism. In other words, respondents’ choices are incentive compatible (Harrison and Florida, 2006).

Table 3-3 Reasons for stating zero willingness to pay

Statement	Number of respondents
Protest zero-bid	
I believe I should not be the one paying for it	5
I don't believe that my payment will be used effectively	5
I don't pay taxes and/or I would prefer another mechanism for paying	1
Total	11
Genuine zero-bid	
I cannot afford to pay	16
I don't think the suggested policies are viable	1
I don't believe there is a need for a restoration project and priorities for public funds should be different	3
Total	20

Stressing the consequentiality of the study also help to obtain incentive compatible choices from respondents (Harrison and Florida, 2006; Herriges et al., 2010). For this reason, we added a paragraph to the survey stating that the study results are policy relevant and “aim to inform decision makers by exploring [...] preferences for managing the [Clyde/Forth/Tay] area”.

A more detailed analysis of the income of protest respondents is developed in figure 3-8. People objecting against the tax *payment vehicle* have the lowest income. Interestingly, the figure also shows that the respondents who consider that it is not their responsibility to fund this policy are positioned on one of the top four levels of income.

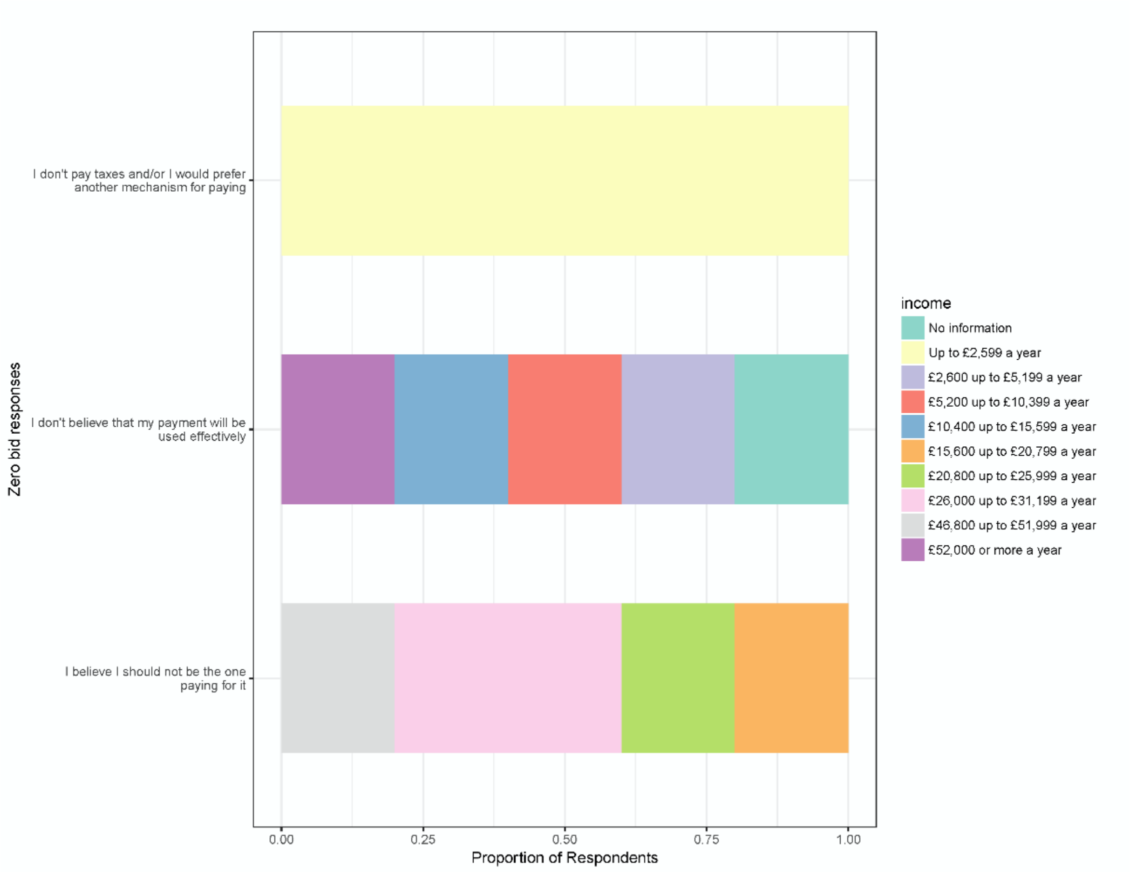


Figure 3-8 Income of protest bid individuals

3.5. Survey sample characteristics

The retrieved data on respondents' postcode (centroid coordinates) was used to geocode their household location in Scotland. The full-length postcode is a relatively precise measure of a respondent's residential location as each postcode unit in the UK covers an average of only 15 properties (Ordnance Survey, 2018). Postcode information was validated with the *assertive.data.uk* package in R (Cotton, 2015) which checks whether the information input contains UK postcodes and thus facilitates the identification of incomplete postcodes, formatting mistakes or fake information. Finally, incomplete postcode information was triangulated with the location of the internet protocol (IP) address collected by Sawtooth Software's CBC/Web system (Sawtooth Software, 2008).

The geographical location of the *final* sample of individuals surveyed is plotted in figure 3-9, and as it can be seen, the obtained sample is non-homogenously distributed across Scotland. The sample point density is higher at the Scottish Central Belt and near the

Aberdeen region (annex 9), which suggest that our sample follow the current trends of population density. Online surveys can be tailored to obtain representatives samples in terms of the socioeconomic characteristics of respondents (Börger, 2016; Lanz and Provins, 2015). Nonetheless, they struggle to obtain geographically homogenously distributed samples and instead are more likely to follow population density or to be restricted to regions with internet access. The latter is not a significant problem in the UK as in the year of 2017 ninety per cent of the households had internet access (Office for National Statistics, 2017a) and 89% of adults were considered to be recent internet users (Office for National Statistics, 2017b).

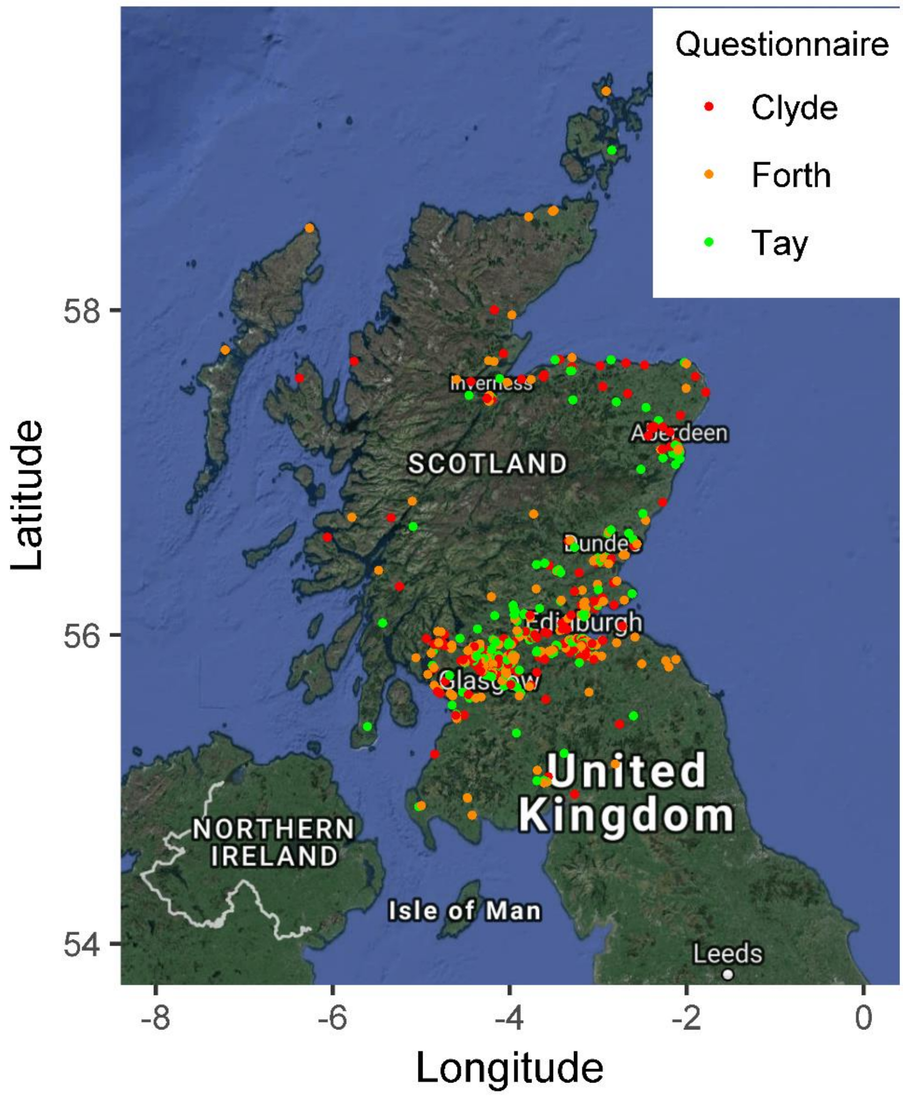


Figure 3-9 Survey respondents map

Table 3-4 summarises the *final* household sample and sub-sample statistics. By means of comparing the sub-datasets, it can be noted that the people answering the Tay questionnaire have higher education, employment and income. Whereas the respondents of the Forth questionnaire have the lowest income and employment rates.

This study used t-tests to determine the representativeness of the *final* sample. The total sample (N=589) is representative of the Scottish population regarding most of the available statistics except age. The percentage of respondents being above 64 years old is significantly different, with 19.69% obtained in our sample vs 16.81% reported in the UK census (Office for National Statistics, 2011).

The self-selection process of the online panel members might incur a small sample bias. Nonetheless, empirical evidence suggests online surveys are a capable method for delivering robust WTP estimations which are not significantly different from those obtained with other surveying methods (Fleming and Bowden, 2009; Olsen, 2009; Windle and Rolfe, 2011). In this sense, the use of existing market research panels did not worsen the sampling coverage error, since our sample is representative of the population on the demographic characteristics (Dillman et al., 2014).

Regarding the rest of the sample socioeconomics characteristics, we found that 31.75% of respondents have residency in the area they valued, and another 52.80% declared they had visited the area for outdoor recreational activities (see the second column in table 3-4). As expected, those living within a catchment area were more likely to make outdoor recreation trips to sites within the area than those living outside. In fact, within the 31.75% of residents, 82.89% declared themselves as visitors of that area. Whereas within the non-residents (68.24%) only 38.80% reported having visited the area.

Finally, table 3-4 presents some statistics on how respondents perceive the environmental status of the study cases compared to ten years ago. It can be seen that the Clyde catchment area had the highest percentage of people perceiving environmental improvements (21.65%), whereas the Tay questionnaire had the highest percentage of people (23.23%) perceiving a worsening on the catchment environmental quality.

Table 3-4 Summary statistics of respondents and their household

Variable	All sites pooled (N=589)	Tay questionnaire (n = 198)	Clyde questionnaire (n = 194)	Forth questionnaire (n = 197)	Scotland's statistics
Income (net, in £ per month)[1]	1765.11 (1153.54)	1837.80 (1211.64)	1790.81 (1194.73)	1666.75 (1041.10)	2249.00[2]
Household size[1]	2.39 (1.84)	2.70 (2.70)	2.33 (1.25)	2.15 (1.05)	2.20[3]
Age (% above 64)	19.69	13.64	13.92	19.29	16.81[4]
Gender (% female)	54.50	58.08	54.64	50.76	51.41[5]
Education (% with university degree and above)	40.41	43.43	36.60	41.12	42.50[5]
Employment (% economically active)	60.27	65.15	61.86	53.81	77.60[5]
Residency in the area (% residents)	31.75	14.65	44.85	36.04	
Visited the area for outdoor recreational activities (% visitors)	52.80	49.49	51.03	57.87	
People perceiving a better environmental status in the area than 10 years ago (% respondents)	18.51	17.17	21.65	16.75	
People perceiving a worse environmental status in the area than 10 years ago (% respondents)	19.86	23.23	15.98	20.30	

Source: Scottish estuarine management Choice Experiment, 2016.
[1] Given are mean and standard deviations in (parenthesis).
[2]Office for National Statistics (2015).
[3]Office for National Statistics (2011).
[4]Office for National Statistics (21013).
[5]Office for National Statistics (2017c).

As explained previously, the DCE survey also included a set of questions to be used as indicators of response quality. Overall, we are confident in the quality of the data collection instrument and the data itself. The highest percentage of respondents (37.35%) declared that when selecting the estuarine management options, their choice was based on the consideration of all (monetary and non-monetary) attributes at the same time. A self-declared quality index (see table 3-5) indicated that the majority of respondents (90.83%) felt confident about how they answered the questionnaire. In addition to this, high percentages of people agreed with the amount of information and the neutrality of it (71.48% and 80.48%, respectively). Finally, more than half of the respondents (67.23%)

perceived that the survey was well targeted and under the half of respondents (47.88%) thought that the policy time frame was not adequate.

Table 3-5 Survey design and data quality statistics

Statement	Agreement in %			
	All sites pooled (N = 589)	Tay questionnaire (n = 198)	Clyde questionnaire (n = 194)	Forth questionnaire (n = 197)
Perceived quality of the survey				
I had enough information for making my choices and understanding the questions	71.48	72.68	68.02	73.74
Information was neutral and not presented in such a way as to influence me	80.48	81.96	76.14	83.33
I am an appropriate individual to be surveyed for this topic	67.23	70.10	68.53	63.13
I believe that the time frame for the project should be shorter than 10 years	47.88	49.48	48.22	45.96
Self-declared data quality				
I am not confident about my answers and choices	9.17	10.10	7.73	9.64
I am confident about my answers and choices	90.83	89.90	92.27	90.36

Six-digit response scale: Strongly disagree, Disagree, Neutral, Agree, Strongly agree, I don’t know. Agreement means agree or strongly agree.

The relative level of importance that respondents attach to each one of the estuarine ES was analysed in this study. Results are presented in table 3-6 and revealed that flood control was ranked as the *most important* ES whereas biodiversity was identified as the *most threatened*. On the other hand, recreation was consistently tagged as the *least important*, and the *least threatened* of all three ES. This information can be useful to cross-validate the DCE results obtained in the analysis chapters, as it can be compared with the estimated weights for each of the ES attributes (see chapter 4).

Table 3-6 Rankings of ES

	Number of people who assigned the label			
Ecosystem Service	Most threatened	Most important	Least threatened	Least important
Flood Control	216	256	116	138
Biodiversity	308	215	76	106
Recreation	80	125	390	344

The total sample (N=589) was used for the first empirical chapter but needed to be reduced for the second and third study in agreement with the analysis requirements. Details about the choice models will be provided in the following chapters, but table 3-7 summarises the samples sizes used in each empirical chapter and the information used to reduce the sample. The three analysis samples were found to be representative of the Scottish population in terms of most of the available statistics, except age. The summary statistics for the first empirical analysis (chapter 4) were already presented in table 3-4. However, the characteristics of the samples used in the chapters 5 and 6 will be summarised in each of these chapters.

Table 3-7 Empirical analysis sample sizes

Empirical chapter	Focus of analysis	Individuals deleted	Sample size
Chapter 4	socioeconomics	protest bid and no postcode information	589
Chapter 5	geographic location	protest bid, no postcode and income information	571
Chapter 6	latent attitudes	protest bid, no postcode and income information, no formed environmental attitude	473

Chapter 4. Socioeconomic effect on preference heterogeneity

4.1. Introduction

This chapter presents the empirical analysis addressing the *Specific objective 1* and answering the questions derived from it. As explained in chapter 1, the first empirical analysis aims to examine how individuals' preferences for policies restoring estuarine ES are influenced by their socioeconomic characteristics. Chapter 2 has already explained why respondent's socioeconomic characteristics could be a potential source of environmental preference heterogeneity. Thus this chapter is particularly interested in analysing whether environmental preferences vary across case study estuaries with different ecological characteristics, and according to the degree on which users make direct use of the ES.

Interactions among ES, such as synergies and trade-offs are likely to be accentuated as the pressures for estuarine natural goods and services increase over time. The development of sustainable ways of management must account for estuarine complexity while considering the values attached to their ES. Understanding the magnitude and distribution of estuarine ES values helps to design mechanisms to mitigate management conflicts and develop optimal management plans.

This chapter, together with the empirical analyses developed in chapters 5 and 6, aims to produce information which could guide policy makers and regulators in designing more efficient and contextualised environmental policies. We estimated both MNL and RPL models with the pooled dataset, as well as separately for each case study estuary. Further to this, we developed a comparative analysis of the relative values *assigned* to estuarine ES, between and within catchment areas. The study embedded in this chapter contributes to the estuarine valuation literature by developing an analysis which considers the complexity of estuarine ecosystems and the benefits they provide to society. This is done by using a *catchment scale* analysis which considers their connectivity with coastal and terrestrial ecosystems. Additionally, we provide a more profound understanding of possible sources of heterogeneity by exploring how environmental preferences and WTP estimates vary across individuals or/and space.

The rest of this chapter is organised as follows. Section 4.2 describes the CM framework used to analyse the choices. Afterwards, in section 4.3, we present and discuss the results of the econometric models, as well as the comparative analysis of welfare estimates. Finally, we present a synthesis of the main findings and discuss their policy implications in section 4.4.

4.2. Empirical analysis

The choice dataset used in this chapter was obtained from a DCE conducted in Scotland in 2016. Details regarding the DCE design and data collection procedures can be found in chapter 3. The analysis of the present chapter uses the choices of a representative sample of 589 individuals (see table 3-4) to estimate society's WTP for improving flood control, recreation and biodiversity in the Clyde, Forth and Tay catchment areas. The summary of the descriptive statistics of the sample and the process for testing its representativeness can be revisited in section 3.5 of the previous chapter.

The analysis presented below has four main sections. The first two sections (4.3.1 and 4.3.2) explore whether there is preference heterogeneity around the mean utility weights, as well as whether individuals with different socioeconomic characteristics have similar preferences for improvements in estuarine ES. The third section (4.3.3), narrows the exploration of preference heterogeneity to focus on two variables i) the study area and ii) the user type. First, we assessed if these characteristics are a significant source of preference heterogeneity for opting out of the *status quo* (using interactions with the ASC). Second, we test whether these characteristics influence preferences for the estuarine ES improvements (using interactions with the attributes). The fourth and final section (4.3.4) analysing the choice data explores whether there is heterogeneity in the WTP estimates for the different ES, among user types and across study cases.

Several specification forms were tested for the following models. We found that the better-fitted models define the utility as a linear function of the attributes of that scenario and the ASC. Table 4-1 describes the coding used in the models. All models were coded and estimated in R software (version 3.3.2). They were estimated with the pooled dataset, as well as with the site-specific datasets and accounted for both, the systematic and stochastic component of preference heterogeneity. Please note that the model output tables have been moved to the end of this chapter for ease of reading.

Table 4-1 Explanation of variable abbreviations and coding

Variable	Explanation
ASC	Constant term (0 = Option1: NO new policy, 1 = Option 2 or 3)
F1	Change in flood control from “increase in flood risk” to “slight reduction in flood risk” (1 = yes, 0 = no)
F2	Change in flood control from “increase in flood risk” to “large reduction in flood risk” (1 = yes, 0 = no)
B1	Change in biodiversity from “decrease in biodiversity” to “slight increase in biodiversity” (1 = yes, 0 = no)
B2	Change in biodiversity from “decrease in biodiversity” to “large increase in biodiversity” (1 = yes, 0 = no)
R1	Change in recreation from “decrease in recreation” to “slight increase in recreation” (1 = yes, 0 = no)
R2	Change in recreation from “decrease in recreation” to “large increase in recreation” (1 = yes, 0 = no)
Cost	Additional council tax payment
Resident	Whether respondent resides in the catchment area (1 = yes, 0 = no)
Visitor	Whether respondent visited the area for outdoor recreational activities in the last 12 months (1 = yes, 0 = no)
Female	Respondent's gender (1 = Female, 0 = Male)
Age	Respondent's age is above the average (1 = yes, 0 = no)
Graduate	Whether respondent has undergraduate and/or postgraduate education (1 = yes, 0 = no)
Income	Respondent's income is above the average for the sample (1 = yes, 0 = no)

4.2.1. Choice modelling

The basis for the analysis of the discrete choice data is the RUM model (McFadden, 1973). According to this model, the total indirect utility U_{int} that an individual derive from alternative i is the sum of its deterministic and random part. The utility of respondent n choosing alternative i in the choice occasion t is given by:

$$U_{int} = V_{int} + \varepsilon_{int} \qquad 4\text{-}1$$

where U_{int} is indirect utility, ε_{int} captures the factors that affect utility but are not observed by the modeller and therefore not included in V_{int}. The deterministic component of utility is given by:

$$V_{int} = f(\beta, x_{int}, z_n) \qquad 4\text{-}2$$

where β_n is a vector of utility weights of respondent n, x_{int} is a vector of attributes of alternative i in choice occasion t, z_n is a vector of measured attributes of respondent n and $\varepsilon_{i,n,t}$ is a random term which is assumed to be independent and identically distributed (IID). Further assuming that a respondent chooses the alternative that maximises their utility, the probability of individual n of choosing alternative i is:

$$P_{int} = \Pr(y_n^t | \cdot) = \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{jnt}}} \qquad 4\text{-}3$$

The equation can be estimated using the MNL model. This model follows the independence of irrelevant alternatives (IIA) assumption which states that the ratio of choice probabilities between any two alternatives in a choice card is not affected by the introduction of removal of additional alternatives (Louviere et al., 2000). Moreover, the MNL assumes homogenous preferences across respondents since it estimates a single (mean) attribute parameter for each choice attribute. Notably, the previously described characteristics of the MNL have been considered relevant limitations and have led to the development of other models.

In this chapter, we also used the RPL[13], which is a model that account for preference heterogeneity by incorporating preference deviation around the attribute means. The utility specification of the RPL model is an extension of equation 4-1 but includes coefficients varying in the population. The equation is rewritten as:

$$U_{int} = (\beta + z_n)\, x_{int} + \varepsilon_{int} \qquad 4\text{-}4$$

The general RPL form is as follows:

$$V_{int} = ASC + \sum \beta_k \cdot \xi_{\mathrm{k}} + \sum \beta_m \cdot Z_{\mathrm{m}} \qquad 4\text{-}5$$

Where the ASC captures the effect of unobserved attributes on the choice, k is the number of attributes and m the number of socioeconomic factors included in the model, if any.

[13] Also referred mixed multinomial logit (MMNL), kernel logit or mixed logit.

In the RPL model, the attribute parameters are assumed to be random, following a specific distribution. Our RPL uses a fixed cost parameter and assumes normally distributed parameters for the ES attributes and the ASC, with mean β and standard deviation σ. The fixed cost coefficient was used to avoid convergence issues and to facilitate the calculation of the implicit prices for the ES attributes (Revelt and Train, 1998; Wielgus et al., 2009). Hence, the conditional choice probability for respondent n choosing alternative i is given by:

$$P_{int} = \Pr(y_n^t | \cdot) = \int_\beta \prod_{t=1}^{Tn} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{ijt}}} f(\beta|\theta) d\beta, \qquad \text{4-6}$$

Finally, the model is estimated by maximum likelihood. The log-likelihood (LL) function for the model is given by $LL(\theta) = \sum_{n=1}^{N} \ln P_{int}$. This expression cannot be solved analytically and simulation-based estimation of the model is used to evaluate P_n at a large number of draws from β, in our case 1,000 Sobol draws. We used this type of draws as they have been found to outperform Halton, modified Latin hypercube sampling, and pseudo-random draws (Czajkowski and Budzinksi, 2017).

The simulated log likelihood of the RPL model is given by:

$$LL(\theta) = \sum_{n=1}^{N} \ln \left[\frac{1}{R} \sum_{r=1}^{R} P_n\left(\beta^{in/\theta}\right) \right] \qquad \text{4-7}$$

where R is the number of draws, $\beta^{in/\theta}$ is a vector of βs obtained in the r-th draw from the distribution $f(\beta|\theta)$ for individual n.

In the RPL model, the parameters of β distribution (θ) are estimated, rather than a vector of β point values as is done in the MNL model.

4.3. Empirical results and discussion

4.3.1. Multinomial logit models

The estimates of the MNL models are reported in table 4-2 together with model fit statistics. This table combines the results of the choice models derived from the (i) pooled dataset, as well as the site-specific choices for the (ii) Clyde, (iii) Forth and (iv) Tay catchment areas. Estimates are stacked to ease the comparison of models, and the respective model from which each estimate is derived is indicated in the column ‘Dataset’.

The MNL models present good explanatory power based on other, comparable studies (ρ^2 ranging from 0.16 to 0.23) since most of the McFadden’s ρ^2 values falling within the range suggested by Hensher and Johnson (1981) as a good fit for choice models. All the attribute coefficients have the expected signs and are found to be significant. The coefficients explaining ES provision levels are positive and often show positive scope effects, which suggest that respondents have a stronger preference for options providing bigger improvements in flood control (F2), biodiversity (B2) and recreational services (R2) than for smaller improvements, respectively. All the ES attributes are positive and significant. The largest coefficients are associated with flood control but are followed closely by biodiversity. Recreation coefficients are smaller by at least a factor of two. The negative sign of the cost attribute in the model indicates that respondents prefer options with lower cost when all other attributes remain constant. The ASC is negative for all cases, but not significant for the Clyde and Forth model. The latter result suggests that on average individuals are supportive of improvements in the supply of ES, as their utility is impacted positively when moving away from the *status quo*.

The MNL model assumes a linear utility function with independent and identically distributed errors with a Gumbel distribution. The IIA assumption implies that the odds of choosing two options depend only on the comparison of their attributes and are not altered by the attributes of any additional alternative. The validity of this assumption was tested on the pooled dataset using the test of Hausman and McFadden (1984), and it was found that IIA is rejected at the 99% level when removing alternative two, but not when removing alternative three. Even though the result is not conclusive, we proceed with the estimation of the RPL models so that we can test if models relaxing this assumption are more appropriate for modelling respondent’s choices for improvements on the provision of estuarine ES.

4.3.2. Random parameter logit models

In order to analyse the stochastic component of preference heterogeneity, we included random coefficients in the utility function (see RPL in table 4-2). In addition to that, the identification of the systematic component of preference heterogeneity is made by using attributes in interaction with socioeconomic variables (see RPL interacted 1 in table 4-2). Both, the simple and interacted models used a normal distribution for the attributes and the ASC (simulated with 1000 Sobol draws), as well as a fixed cost parameter.

The log-likelihood statistics indicated that the model with interactions had a better overall fit ($LL_{\text{RPL interacted 1}} = -2744.44$) than the other two simpler models ($LL_{RPL} = -2755.44, LL_{MNL} = -3140.18$). To test whether the differences in log-likelihood are due to the addition of variables we followed Hess and Daly (2010) suggestion and developed a pairwise comparison of the models presented in table 4-2. The model comparison uses the log-likelihood ratio which test the null hypothesis $D = -2(lnL_R - lnL_U)$, with degrees of freedom equal to the difference in the number of parameters between the compared models. The likelihood ratio test indicated that at the 5 percent significance level the RPL interacted 1 model fit the data best.[14]. The RPL interacted 1 model also remains as the better-fitted model when using other model comparison measures (lowest AIC and BIC statistic, with 5530.88 and 5660.45, respectively).

All the estimated RPL models present significant coefficients and have the expected signs for all attributes. The ASC becomes significant in the simple RPL models but is not significant for the models which also account for the systematic preference heterogeneity (RPL interacted 1). The ranking of attributes and the scope effect found in the MNL models persist. Lastly, the standard deviations followed the same patterns of significance in both models, the simple and interacted RPL, which reveals the presence of significant random preference heterogeneity for almost all attribute levels, except the smaller improvements in biodiversity (B1) and recreation (R1).

The model with interactions (RPL interacted 1) is used to understand the drivers of preference heterogeneity for improvements in the provision of estuarine ES. In this model, we tested the significance of the full set of socioeconomic variables. The model estimated with the pooled dataset presents negative and significant coefficients for

[14] The critical value for the log-likelihood ratio comparing MNL and RPL was $(p = 0.05) = 7.161458E - 162$. The critical value for the log-likelihood ratio comparing the RPL models (without and with interactions) was $(p = 0.05) = 0.00074$.

visitors, female and age. In other words, visitors, females and older respondents are significantly more likely to choose scenarios improving estuarine ES provision. The income coefficient is negative but fails to reach significance, which indicates that this variable is not a significant driver of preference heterogeneity.

The previous results are in line with suggestions in the literature regarding the effects of age and gender as significant sources of preference heterogeneity (Andreopoulos et al., 2015a; Börger and Hattam, 2017; Botzen et al., 2012; Hanley et al., 2007). Previous valuation studies have also found that visiting the area of interest is also a relevant factor for seeking for environmental improvement (Birol et al., 2009; Samdin and Khairil, 2010; Zander et al., 2010). Interestingly, our results indicate that the level of education (having a graduate degree) is not a significant factor for having preferences for ES level improvements. However, previous studies have shown that the results associated with the education variable (or education effect) often depend on the user type (see Jobstvogt et al., 2014) or the environmental good to be valued (Hanley et al., 2007).

Table 4-2 MNL and RPL estimates for ES improvements

		MNL			RPL						RPL interacted 1					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1	All	1.12	***	0.06	1.66	***	0.10	0.61	***	0.14	1.66	***	0.10	0.60	***	0.14
	Clyde	1.12	***	0.11	1.75	***	0.19	0.65	**	0.24	1.73	***	0.18	0.59	*	0.25
	Forth	1.08	***	0.11	1.69	***	0.19	0.70	**	0.25	1.68	***	0.18	0.67	*	0.25
	Tay	1.19	***	0.12	1.57	***	0.17	0.53	+	0.25	1.59	***	0.17	0.60	*	0.24
F2	All	1.39	***	0.07	2.12	***	0.13	1.15	***	0.12	2.12	***	0.13	1.13	***	0.12
	Clyde	1.39	***	0.13	2.25	***	0.24	1.24	***	0.21	2.22	***	0.23	1.21	***	0.21
	Forth	1.21	***	0.12	1.92	***	0.23	1.30	***	0.23	1.91	***	0.23	1.31	***	0.23
	Tay	1.64	***	0.13	2.24	***	0.22	0.85	***	0.20	2.27	***	0.22	0.87	***	0.20
B1	All	1.00	***	0.07	1.68	***	0.11	0.29		0.22	1.67	***	0.11	0.24		0.26
	Clyde	1.08	***	0.13	1.88	***	0.21	0.26		0.37	1.86	***	0.21	0.25		0.42
	Forth	0.92	***	0.13	1.64	***	0.21	0.63	**	0.23	1.63	***	0.20	0.60	*	0.24
	Tay	1.01	***	0.13	1.50	***	0.19	0.01		0.28	1.52	***	0.19	0.01		0.31
B2	All	1.10	***	0.08	1.81	***	0.12	0.80	***	0.12	1.81	***	0.12	0.79	***	0.12
	Clyde	0.94	***	0.13	1.70	***	0.21	0.81	***	0.22	1.68	***	0.21	0.57	*	0.23
	Forth	1.07	***	0.13	1.87	***	0.23	0.81	***	0.23	1.86	***	0.22	0.83	***	0.23
	Tay	1.32	***	0.14	1.89	***	0.21	0.86	***	0.18	1.92	***	0.21	0.88	***	0.18
R1	All	0.38	***	0.06	0.64	***	0.08	0.02		0.21	0.64	***	0.08	0.02		0.21
	Clyde	0.37	***	0.10	0.69	***	0.14	0.02		0.35	0.68	***	0.14	0.04		0.32
	Forth	0.42	***	0.10	0.78	***	0.15	0.00		0.29	0.77	***	0.14	0.06		0.27
	Tay	0.34	***	0.10	0.50	***	0.13	0.13		0.68	0.50	***	0.13	0.20		0.38
R2	All	0.40	***	0.06	0.63	***	0.08	0.61	***	0.20	0.63	***	0.08	0.60	***	0.13
	Clyde	0.44	***	0.10	0.72	***	0.15	0.67	**	0.20	0.71	***	0.15	0.65	**	0.23

		MNL			RPL						RPL interacted 1					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Forth	0.29	**	0.10	0.58	***	0.15	0.80	***	0.20	0.57	***	0.15	0.80	***	0.20
	Tay	0.46	***	0.10	0.62	***	0.12	0.31		0.20	0.63	***	0.13	0.30		0.36
Cost	All	-0.01	***	0.00	-0.01	***	0.00	-	-	-	-0.01	***	0.00	-	-	-
	Clyde	-0.01	***	0.00	-0.02	***	0.00	-	-	-	-0.02	***	0.00	-	-	-
	Forth	-0.01	***	0.00	-0.01	***	0.00	-	-	-	-0.01	***	0.00	-	-	-
	Tay	-0.01	***	0.00	-0.01	***	0.00	-	-	-	-0.01	***	0.00	-	-	-
ASC	All	-0.23	*	0.09	-1.86	***	0.29	3.29	***	0.27	0.05		0.53	3.12	***	0.26
	Clyde	-0.03		0.16	-1.81	***	0.54	3.86	***	0.53	0.84		0.96	3.50	***	0.49
	Forth	-0.29	+	0.16	-1.83	***	0.51	3.40	***	0.48	0.72		0.89	3.07	***	0.46
	Tay	-0.40	*	0.17	-1.88	***	0.46	2.79	***	0.41	-1.15		0.85	2.63	***	0.39
*ASC*Resident*	All	-		-	-		-	-		-	0.57		0.45	-		-
	Clyde	-		-	-		-	-		-	0.61		0.83	-		-
	Forth	-		-	-		-	-		-	0.38		0.73	-		-
	Tay	-		-	-		-	-		-	-0.01		0.87	-		-
*ASC*Visitor*	All	-		-	-		-	-		-	-1.02	*	0.42	-		-
	Clyde	-		-	-		-	-		-	-1.02		0.82	-		-
	Forth	-		-	-		-	-		-	-1.37	+	0.76	-		-
	Tay	-		-	-		-	-		-	-0.77		0.60	-		-
*ASC*Female*	All	-		-	-		-	-		-	-0.86	*	0.39	-		-
	Clyde	-		-	-		-	-		-	-0.67		0.71	-		-
	Forth	-		-	-		-	-		-	-1.25	+	0.66	-		-
	Tay	-		-	-		-	-		-	-0.37		0.63	-		-
*ASC*Age*	All	-		-	-		-	-		-	-1.15	**	0.39	-		-
	Clyde	-		-	-		-	-		-	-1.89	*	0.76	-		-

		MNL		RPL				RPL interacted 1				
Attribute	Dataset	Coeff. (Mean)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)	S.E.
	Forth	-	-	-	-	-	-	-1.67	*	0.66	-	-
	Tay	-	-	-	-	-	-	-0.26		0.62	-	-
*ASC*Graduate*	All	-	-	-	-	-	-	0.16		0.39	-	-
	Clyde	-	-	-	-	-	-	0.31		0.74	-	-
	Forth	-	-	-	-	-	-	-0.97		0.77	-	-
	Tay	-	-	-	-	-	-	1.05	+	0.60	-	-
*ASC*Income*	All	-	-	-	-	-	-	-0.03	+	0.01	-	-
	Clyde	-	-	-	-	-	-	-0.06	*	0.03	-	-
	Forth	-	-	-	-	-	-	0.01		0.03	-	-
	Tay	-	-	-	-	-	-	-0.02		0.02	-	-
Log-likelihood	All	-3140.18		-2755.44				-2744.44				
	Clyde	-1062.12		-899.20				-893.11				
	Forth	-1059.96		-924.21				-917.20				
	Tay	-993.40		-911.36				-908.02				
Observations	All	3534.00		3534.00				3534.00				
	Clyde	1164.00		1164.00				1164.00				
	Forth	1182.00		1182.00				1182.00				
	Tay	1188.00		1188.00				1188.00				
Adjusted rho-sq	All	0.19		0.29				0.29				
	Clyde	0.16		0.29				0.29				
	Forth	0.18		0.28				0.28				
	Tay	0.23		0.29				0.29				
AIC	All	6296.36		5540.89				5530.88				

		MNL		RPL				RPL interacted 1			
Attribute	Dataset	Coeff. (Mean)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.
	Clyde	2140.25		1828.39				1828.23			
	Forth	2135.91		1954.54				1876.40			
	Tay	2002.80		1852.71				1858.03			
BIC	All	6345.72		5633.44				5660.45			
	Clyde	2180.72		1904.29				1934.48			
	Forth	2176.51		1954.26				1982.97			
	Tay	2043.44		1928.91				1964.71			

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

4.3.3. Effects of use and location

In this section, we extend the heterogeneity analysis by including two factors into the comparative analysis. The first factor is the ‘study site’ for which people were asked to value changes in estuarine ES provision. This analysis was applied to the pooled and site-specific datasets in order to obtain the general and site-specific estimates which are presented in the rows of each output table. The second factor is the ‘user type’ for which we analyse the following three categories: i) being a resident of the area, ii) visiting the area for outdoor recreational activities and ii) being both a resident and a visitor of the area. We defined these categories so that they describe the extent to which respondents have a use or non-use value for improvements in estuarine ES provision. Moreover, these categories allow us to compare test whether use values for residents are different from use values for visitors.

It has to be noted that the categories are not mutually exclusive and therefore it was necessary to study them in independent models. Moreover, the sample numbers were not high enough for using data subsets. Thus we opted for using these variables in interaction with the utility function components. As we are dealing with three binary variables and we do not have *a priori* reason to know which variable to choose for the dummy analysis, it was sensible to impose the mirror condition and include both dummies in the model. Doing so allows us to obtain user-specific WTP estimates to be compared in section 4.3.4.

Several RPL models with interactions were estimated (using the pooled and site-specific datasets) to understand the heterogeneity regarding the preferences for change and the WTP estimates. The first set of models (see table 4-3) aims to compare the preferences for opting out of the *status quo* and includes the ‘user type’ variables in interaction with the ASC. The second set of models (see table 4-4, table 4-5 and table 4-6) use the ‘user type’ variables in interaction with the ES attributes and is used to estimate the user-specific attribute coefficients. These models are subsequently used to explore the relative differences in WTP within and between catchment areas.

The table 4-3 present the three ‘ASC interacted models’ and shows that the magnitude and significance patterns of the attributes and standard deviation coefficients remain the same. All attributes exhibit the expected significant coefficients. The standard deviation of the small improvements in biodiversity (B1) and recreation (R1) remain insignificant. Besides including the *general ASC*, the three interacted RPL models in table 4-3 have additional *group-specific ASC*. The general ASC is significantly negative in the three models and indicates a general preference for change or to avoid the *status quo*. Regarding the *group-specific ASC*, almost all case studies present negative coefficients, but the constant was only significant for the visitors model estimated with the pooled dataset (see row named ‘All’ in the RPL interacted 3 model). The significance of the visitor variable indicates that unlike residents, visitors are significantly more likely to have a preference for opting out of the *status quo.* The reduction of the sample size used to estimate the subset models might explain the loss of significance in the ‘visitor’ variables. However, in order to asses whether the ‘visitor’ effect is significant or not, we proceed to develop a comparative analysis of WTP estimates in the following section.

Further analysis of the magnitude of the *group-specific ASC* coefficients leads to two conclusions. Firstly, visitors exhibit stronger preferences for change as their ASC coefficients are consistently more negative when compared to the rest of ‘user-type interacted models’ (see columns in table 4-3). Secondly, we found less negative ASC coefficients for the Clyde sub-sample when compared to the other case studies, meaning that the lowest preference for change is associated to this area (see rows in table 4-3). The outputs from table 4-3 indicate that there is preference heterogeneity for opting out of the *status quo* and suggest that visiting the area is a significant source of preference heterogeneity.

Tables 4-4 to 4-6 display the outputs for the remaining RPL models which include attributes in interactions with the ‘user type’ variables to test for dissimilar preferences associated with the ES attributes. The attribute coefficients for the three models present the expected signs and are in most of the cases significantly different from zero. Nonetheless, this slight reduction of significance in the attributes parameters is associated with the substantial reduction in sample sizes, as fewer respondents fall into this category. In comparison to the previous models, there is also a reduction on the numbers and levels of significance related to the standard deviation coefficients. In fact, the presence of still significant standard deviation coefficients indicates that even when accounting for the systematic heterogeneity explained by the ‘user type’, there is still a residual component of random heterogeneity that is not explained in the models.

The RPL using attribute interactions (RPL interacted 5-7) revealed that the ‘study‘user type’ variables have an effect on individuals’ preferences for improving estuarine ES provision levels. The significance of this effect can be assessed by means of re-specifying these models (see annex 10) so that they include n-1 dummies. Annex 10 indicates that ‘residents’, ‘visitors’ and ‘both resident and visitors’are associated with a higher probability of choosing the management alternatives delivering estuarine ES. The significant effect of the ‘study site’ variables cannot be assessed directly from any of the previously mentioned models, thus we use a comparative analysis of WTP estimates in the following section to obtain further insights of this.

Further exploration of the final three model outputs (RPL interacted 5-7) reveals that there is no general pattern of higher WTP estimates associated with specific ‘user types’, meaning that all users care about different ES. The groups of respondents identified as ‘non-residents’, ‘visitors’ and ‘neither residents nor visitors’ have the same ES preference ranking identified in previous results, with flood control as the most preferred estuarine ES and recreation as the least preferred. However, the group of ‘non-visitors’, ‘residents’ and ‘both resident and visitors’ have a higher ranking for biodiversity and declared flood control as the second most preferred ES. In this sense, being a user does not necessarily impact the absolute values of ES a positive way, but influences the relative preferences and determines respondent’s priorities for restoring estuarine ES in Scotland.

Interestingly we found that those groups assigning more passive values (e.g., ‘non-residents’ and ‘neither’) prioritised flood control over the rest of ES. Even though this result seems to be unexpected at first glance, flood control benefits are provided even outside the catchment area in which they are generated. Thus this group of people might be expressing the ‘indirect-use value’ they assign to flood control (Mehvar et al., 2018).

On the other hand, the direct users of estuarine ES (e.g., ‘residents’ and ‘both’) perceived biodiversity as the most relevant ES. Those respondents might be expressing the existence or bequest value of biodiversity (Mehvar et al., 2018). Alternatively, this outcome might indicate, that to a certain extent; respondents perceive biodiversity’s capacity to be the sustenance of all other services (Balvanera et al., 2006; Hector and Bagchi, 2007). In fact, 73.78% of the respondents agreed with the statement *“biodiversity is essential for the production of goods such as food or fuel”* (see details in table 6-2).

The comparison of the fit to the data of the six RPL interacted models presented in this section (see tables 4-3 to 4-6) cannot be done by means of the likelihood ratio test, as they are not nested models. Instead, we used the Bayesian information criterion (BIC) statistic to compare models. This statistic was selected over the Akaike information criterion (AIC) because it penalises models with more parameters more strongly (Burnham and Anderson, 2002). The six models analysed in this section (RPL interacted 2-7) have higher BIC statistics and do not outperform the RPL interacted 1 model, however, they are used in this study to answer the research questions that derive from the *Specific objective 1*.

Table 4-3 User type ASC interacted RPL estimates for ES improvement

		RPL interacted 2						RPL interacted 3						RPL interacted 4					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1	All	1.66	***	0.10	0.61	***	0.14	1.67	***	0.10	0.63	***	0.14	1.67	***	0.10	0.64	***	0.14
	Clyde	1.73	***	0.18	0.60	*	0.25	1.74	***	0.18	0.63	**	0.24	1.74	***	0.19	0.64	**	0.24
	Forth	1.69	***	0.19	0.70	**	0.24	1.68	***	0.18	0.69	**	0.24	1.70	***	0.19	0.71	**	0.24
	Tay	1.58	***	0.17	0.56	*	0.24	1.59	***	0.17	0.57	*	0.24	1.59	***	0.17	0.59	*	0.24
F2	All	2.12	***	0.13	1.14	***	0.12	2.13	***	0.13	1.16	***	0.12	2.13	***	0.13	1.15	***	0.12
	Clyde	2.22	***	0.23	1.22	***	0.21	2.23	***	0.23	1.24	***	0.21	2.22	***	0.24	1.24	***	0.21
	Forth	1.91	***	0.23	1.30	***	0.23	1.90	***	0.23	1.28	***	0.22	1.94	***	0.23	1.31	***	0.23
	Tay	2.27	***	0.22	0.87	***	0.20	2.26	***	0.22	0.86	***	0.20	2.27	***	0.22	0.87	***	0.20
B1	All	1.68	***	0.11	0.24		0.29	1.69	***	0.11	0.28		0.24	1.69	***	0.11	0.28		0.23
	Clyde	1.86	***	0.20	0.21		0.45	1.87	***	0.21	0.01		NA	1.87	***	0.21	0.09		1.67
	Forth	1.64	***	0.21	0.63	**	0.24	1.64	***	0.21	0.62	*	0.24	1.65	***	0.21	0.63	*	0.24
	Tay	1.52	***	0.19	0.01		0.29	1.52	***	0.19	0.01		0.30	1.52	***	0.19	0.02		0.30
B2	All	1.81	***	0.12	0.80	***	0.12	1.82	***	0.12	0.81	***	0.12	1.82	***	0.12	0.81	***	0.12
	Clyde	1.68	***	0.21	0.57	*	0.23	1.69	***	0.21	0.61	*	0.22	1.69	***	0.21	0.59	*	0.22
	Forth	1.87	***	0.22	0.84	***	0.22	1.86	***	0.22	0.82	***	0.23	1.87	***	0.23	0.83	***	0.23
	Tay	1.91	***	0.22	0.87	***	0.18	1.91	***	0.21	0.86	***	0.18	1.91	***	0.22	0.87	***	0.18
R1	All	0.64	***	0.08	0.01		0.22	0.64	***	0.08	0.03		0.22	0.65	***	0.08	0.00		0.22
	Clyde	0.68	***	0.14	0.06		0.35	0.68	***	0.14	0.03		0.29	0.68	***	0.14	0.03		0.32
	Forth	0.77	***	0.15	0.01		0.27	0.77	***	0.14	0.03		0.27	0.78	***	0.15	0.04		0.28
	Tay	0.50	***	0.13	0.16		0.51	0.50	***	0.13	0.21		0.39	0.51	***	0.13	0.18		0.44
R2	All	0.64	***	0.08	1.14	***	0.12	0.64	***	0.08	1.16	***	0.12	0.64	***	0.08	1.15	***	0.12
	Clyde	0.71	***	0.15	1.22	***	0.21	0.72	***	0.15	1.24	***	0.21	0.71	***	0.15	1.24	***	0.21
	Forth	0.58	***	0.15	1.30	***	0.23	0.58	***	0.15	1.28	***	0.22	0.59	***	0.15	1.31	***	0.23
	Tay	0.63	***	0.13	0.87	***	0.20	0.62	***	0.12	0.86	***	0.20	0.63	***	0.13	0.87	***	0.20
Cost	All	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-
	Clyde	-0.02	***	0.00	-		-	-0.02	***	0.00	-		-	-0.02	***	0.00	-		-
	Forth	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-
	Tay	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-	-0.01	***	0.00	-		-
ASC	All	-1.91	***	0.32	3.28	***	0.27	-1.42	***	0.33	3.28	***	0.26	-1.82	***	0.31	3.33	***	0.27
	Clyde	-1.95	**	0.63	3.78	***	0.27	-1.46	*	0.60	3.81	***	0.27	-1.67	**	0.58	3.78	***	0.27

Attribute	Dataset	RPL interacted 2				RPL interacted 3				RPL interacted 4			
		Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.
	Forth	-1.85 **	0.58	3.34 ***	0.27	-1.21 +	0.61	3.34 ***	0.27	-1.86 **	0.57	3.37 ***	0.27
	Tay	-1.80 ***	0.47	2.72 ***	0.27	-1.44 **	0.51	2.71 ***	0.27	-1.80 ***	0.46	2.71 ***	0.27
*ASC*Resident*	All	0.16	0.41	-	-	-	-	-	-	-	-	-	-
	Clyde	0.37	0.73	-	-	-	-	-	-	-	-	-	-
	Forth	-0.03	0.72	-	-	-	-	-	-	-	-	-	-
	Tay	-0.25	0.90	-	-	-	-	-	-	-	-	-	-
*ASC*Visitor*	All	-	-	-	-	-0.83 *	0.38	-	-	-	-	-	-
	Clyde	-	-	-	-	-0.73	0.65	-	-	-	-	-	-
	Forth	-	-	-	-	-1.11	0.67	-	-	-	-	-	-
	Tay	-	-	-	-	-0.82	0.58	-	-	-	-	-	-
*ASC*Visitor*Resident*	All	-	-	-	-	-	-	-	-	-0.10	0.44	-	-
	Clyde	-	-	-	-	-	-	-	-	-0.37	0.79	-	-
	Forth	-	-	-	-	-	-	-	-	-0.05	0.70	-	-
	Tay	-	-	-	-	-	-	-	-	-0.38	0.95	-	-
Log-likelihood	All	-2755.34				-2752.05				-2754.54			
	Clyde	-899.89				-898.99				-899.51			
	Forth	-924.25				-923.14				-924.12			
	Tay	-910.74				-909.82				-910.51			
Observations	All	3534.00				3534.00				3534.00			
	Clyde	1164.00				1164.00				1164.00			
	Forth	1182.00				1182.00				1182.00			
	Tay	1188.00				1188.00				1188.00			
Adjusted rho-sq	All	0.29				0.29				0.29			
	Clyde	0.28				0.28				0.28			
	Forth	0.28				0.28				0.28			
	Tay	0.29				0.29				0.29			
AIC	All	5542.67				5536.09				5541.08			
	Clyde	1831.77				1829.97				1831.01			
	Forth	1880.49				1878.29				1880.24			
	Tay	1853.48				1851.64				1853.02			
BIC	All	5641.39				5634.81				5639.81			
	Clyde	1912.73				1910.93				1911.97			

		RPL interacted 2				RPL interacted 3				RPL interacted 4			
Attribute	Dataset	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.
	Forth	1961.69				1959.49				1961.44			
	Tay	1934.76				1932.92				1934.30			

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

Table 4-4 Resident attribute interacted RPL estimates for ES improvement

		RPL interacted 5					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Resident	All	1.19	***	0.16	0.70	**	0.22
	Clyde	1.01	***	0.22	0.56	+	0.32
	Forth	1.57	***	0.28	0.54		0.47
	Tay	1.49	*	0.57	1.57	**	0.64
F2*Resident	All	1.33	***	0.19	1.19	***	0.20
	Clyde	1.19	***	0.26	0.86	**	0.27
	Forth	1.38	***	0.36	1.52	***	0.38
	Tay	2.57	**	0.72	2.38	**	0.90
B1*Resident	All	1.43	***	0.18	0.41	+	0.27
	Clyde	1.37	***	0.26	0.23		0.56
	Forth	1.55	***	0.32	0.41		0.58
	Tay	1.90	***	0.56	0.21		0.75
B2*Resident	All	1.42	***	0.19	0.35		0.32
	Clyde	1.07	***	0.25	0.02		0.51
	Forth	1.69	***	0.35	0.81	*	0.39
	Tay	2.90	***	0.70	0.09		0.74
R1*Resident	All	0.71	***	0.13	0.22		0.39
	Clyde	0.63	**	0.19	0.02		0.51
	Forth	0.98	***	0.24	0.02		0.67
	Tay	0.22		0.44	0.85		0.70
R2*Resident	All	0.96	***	0.14	0.68	**	0.21
	Clyde	1.02	***	0.20	0.32		0.43
	Forth	0.89	**	0.27	0.98	**	0.31
	Tay	1.14	*	0.44	1.02		0.79
Cost *Resident	All	-0.01	***	0.00	-		-
	Clyde	-0.01	***	0.00	-		-
	Forth	-0.01	**	0.00	-		-
	Tay	-0.02	**	0.01	-		-
F1*Non-Resident	All	1.92	***	0.13	0.62	***	0.17
	Clyde	2.55	***	0.31	0.82	*	0.36
	Forth	1.85	***	0.24	0.78	*	0.30
	Tay	1.63	***	0.17	0.45		0.27
F2*Non-Resident	All	2.55	***	0.17	1.11	***	0.15
	Clyde	3.47	***	0.43	1.75	***	0.35
	Forth	2.28	***	0.29	1.16	***	0.29
	Tay	2.28	***	0.22	0.71	**	0.22
B1*Non-Resident	All	1.83	***	0.14	0.30		0.28
	Clyde	2.46	***	0.32	0.41		0.55
	Forth	1.76	***	0.26	0.71	*	0.30
	Tay	1.50	***	0.19	0.02		0.32
B2*Non-Resident	All	2.04	***	0.15	0.99	***	0.14
	Clyde	2.37	***	0.34	1.16	**	0.31
	Forth	2.04	***	0.29	0.96	***	0.28
	Tay	1.84	***	0.22	0.93	***	0.19
R1*Non-Resident	All	0.61	***	0.09	0.01		0.21
	Clyde	0.76	***	0.21	0.04		0.44
	Forth	0.66	***	0.18	0.02		0.26
	Tay	0.50	***	0.13	0.03		0.53
R2*Non-Resident	All	0.50	***	0.10	0.56	***	0.16
	Clyde	0.41	+	0.22	0.69	*	0.36

RPL interacted 5

Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Forth	0.44	*	0.18	0.71	*	0.27
	Tay	0.54	***	0.13	0.16		0.55
Cost *Non-Resident	All	-0.02	***	0.00	-		-
	Clyde	-0.03	***	0.00	-		-
	Forth	-0.02	***	0.00	-		-
	Tay	-0.01	***	0.00	-		-
ASC	All	-1.83	***	0.29	3.28	***	0.26
	Clyde	-1.78	***	0.53	3.78	***	0.52
	Forth	-1.84	***	0.52	3.37	***	0.48
	Tay	-1.87	***	0.46	2.78	***	0.40
Log-likelihood	All	-2726.30					
	Clyde	-871.78					
	Forth	-916.38					
	Tay	-900.72					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.30					
	Forth	0.27					
	Tay	0.29					
AIC	All	5508.61					
	Clyde	1799.57					
	Forth	1888.76					
	Tay	1857.45					
BIC	All	5681.37					
	Clyde	1941.24					
	Forth	2030.86					
	Tay	1999.69					

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

Table 4-5 Visitor attribute interacted RPL estimates for ES improvement

	RPL interacted 6						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Visitor	All	1.58	***	0.13	0.51	*	0.20
	Clyde	1.40	***	0.23	0.71	*	0.30
	Forth	1.83	***	0.26	0.69	+	0.34
	Tay	1.80	***	0.24	0.12		0.62
F2*Visitor	All	1.98	***	0.17	1.23	***	0.16
	Clyde	1.80	***	0.29	1.16	***	0.28
	Forth	1.96	***	0.33	1.70	***	0.35
	Tay	2.51	***	0.30	0.96	***	0.26
B1*Visitor	All	1.46	***	0.14	0.38		0.24
	Clyde	1.51	***	0.26	0.01		0.46
	Forth	1.54	***	0.29	0.94	**	0.31
	Tay	1.51	***	0.24	0.03		0.46
B2*Visitor	All	1.66	***	0.16	0.69	***	0.17
	Clyde	1.35	***	0.26	0.30		0.40
	Forth	1.91	***	0.32	1.08	***	0.33
	Tay	2.00	***	0.29	0.83	**	0.25
R1*Visitor	All	0.74	***	0.10	0.03		0.53
	Clyde	0.68	***	0.19	0.04		0.55
	Forth	1.00	***	0.21	0.11		0.44
	Tay	0.66	***	0.17	0.27		0.47
R2*Visitor	All	0.84	***	0.11	0.52	**	0.18
	Clyde	0.91	***	0.19	0.35		0.45
	Forth	0.82	**	0.22	0.99	**	0.29
	Tay	0.95	***	0.17	0.06		0.57
Cost *Visitor	All	-0.01	***	0.00	-		-
	Clyde	-0.01	***	0.00	-		-
	Forth	-0.01	***	0.00	-		-
	Tay	-0.01		0.00	-		-
F1*Non-Visitor	All	1.76	***	0.15	0.68	***	0.20
	Clyde	2.26	***	0.29	0.44		0.47
	Forth	1.62	***	0.25	0.69	*	0.33
	Tay	1.51	***	0.24	0.89	**	0.31
F2*Non-Visitor	All	2.27	***	0.18	1.01	***	0.18
	Clyde	2.87	***	0.38	1.44	***	0.33
	Forth	1.93	***	0.28	0.69	+	0.36
	Tay	2.15	***	0.29	0.81	*	0.33
B1*Non-Visitor	All	1.92	***	0.16	0.07		0.44
	Clyde	2.37	***	0.32	0.63	+	0.38
	Forth	1.85	***	0.28	0.01		0.49
	Tay	1.62	***	0.26	0.03		0.45
B2*Non-Visitor	All	1.97	***	0.17	0.93	***	0.16
	Clyde	2.17	***	0.34	1.07	**	0.32
	Forth	1.97	***	0.30	0.72	*	0.29
	Tay	1.90	***	0.29	1.03	***	0.26
R1*Non-Visitor	All	0.51	***	0.11	0.04		0.22
	Clyde	0.70	***	0.22	0.07		0.39
	Forth	0.51	**	0.19	0.00		0.26
	Tay	0.34	+	0.18	0.24		0.48
R2*Non-Visitor	All	0.38	**	0.12	0.65	***	0.18
	Clyde	0.45	+	0.23	0.77	*	0.36

RPL interacted 6

Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Forth	0.39	+	0.20	0.61	+	0.31
	Tay	0.28		0.18	0.48		0.37
Cost *Non-Visitor	All	-0.02	***	0.00	-		-
	Clyde	-0.02	***	0.00	-		-
	Forth	-0.02	***	0.00	-		-
	Tay	-0.01		0.00	-		-
ASC	All	-1.90	***	0.30	3.33	***	0.27
	Clyde	-1.97	***	0.57	3.98	***	0.53
	Forth	-1.83	***	0.52	3.37	***	0.48
	Tay	-1.79		0.45	2.73		0.39
Log-likelihood	All	-2742.40					
	Clyde	-889.11					
	Forth	-915.65					
	Tay	-903.04					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.28					
	Forth	0.27					
	Tay	0.29					
AIC	All	5540.79					
	Clyde	1834.23					
	Forth	1887.30					
	Tay	1862.07					
BIC	All	5713.56					
	Clyde	1975.90					
	Forth	2029.40					
	Tay	2004.32					

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

Table 4-6 Resident and visitor attribute interacted RPL estimates for ES improvement

	RPL interacted 7						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Visitor*Resident	All	1.14	***	0.17	0.68	*	0.26
	Clyde	0.88	***	0.23	0.53		0.38
	Forth	1.60	***	0.32	0.44		0.61
	Tay	2.07		1.16	1.37		1.10
F2*Visitor*Resident	All	1.29	***	0.22	1.26	***	0.23
	Clyde	1.20	***	0.29	0.89	**	0.29
	Forth	1.23	**	0.42	1.69	**	0.45
	Tay	2.67		1.28	2.59	+	1.28
B1*Visitor*Resident	All	1.38	***	0.21	0.33		0.35
	Clyde	1.43	***	0.30	0.01		0.59
	Forth	1.36	**	0.37	0.35		0.63
	Tay	2.04		1.18	0.26		1.04
B2*Visitor*Resident	All	1.43	***	0.22	0.57	*	0.25
	Clyde	1.14	***	0.29	0.28		0.43
	Forth	1.52	**	0.41	0.99	*	0.44
	Tay	3.46	+	1.57	0.06		0.81
R1*Visitor*Resident	All	0.70	***	0.15	0.23		0.53
	Clyde	0.62	**	0.22	0.20		0.66
	Forth	0.92	**	0.28	0.23		0.84
	Tay	0.37		0.57	1.15		0.86
R2*Visitor*Resident	All	0.98	***	0.16	0.63	*	0.25
	Clyde	1.03	***	0.22	0.29		0.50
	Forth	0.87	*	0.31	1.15	*	0.39
	Tay	1.54		0.81	1.02		0.87
Cost *Visitor*Resident	All	-0.01	***	0.00	-		-
	Clyde	-0.01	**	0.00	-		-
	Forth	-0.01	*	0.00	-		-
	Tay	-0.02		0.01	-		-
F1*Non-Visitor*Non-Resident	All	1.77	***	0.16	0.76	***	0.21
	Clyde	2.35	***	0.34	0.46		0.58
	Forth	1.67	***	0.29	0.81	**	0.35
	Tay	1.39	***	0.23	0.78	*	0.31
F2*Non-Visitor*Non-Resident	All	2.33	***	0.20	1.06	***	0.20
	Clyde	3.35	***	0.47	1.69	***	0.40
	Forth	1.97	***	0.32	0.79	+	0.41
	Tay	1.91	***	0.27	0.73	*	0.34
B1*Non-Visitor*Non-Resident	All	1.92	***	0.18	0.01		0.47
	Clyde	2.71	***	0.38	0.60		0.44
	Forth	1.77	***	0.32	0.01		0.46
	Tay	1.37	***	0.26	0.02		0.45
B2*Non-Visitor*Non-Resident	All	2.03	***	0.19	1.02	***	0.18
	Clyde	2.54	***	0.41	1.37	***	0.35
	Forth	1.90	***	0.34	0.83	**	0.31
	Tay	1.69	***	0.29	0.97	***	0.26
R1*Non-Visitor*Non-Resident	All	0.46	***	0.12	0.03		0.23
	Clyde	0.72	**	0.25	0.09		0.41
	Forth	0.41	*	0.21	0.02		0.30
	Tay	0.24		0.18	0.02		0.50
R2*Non-Visitor*Non-Resident	All	0.32	*	0.12	0.61	**	0.20

	RPL interacted 7						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Clyde	0.31		0.27	0.74	+	0.41
	Forth	0.30		0.23	0.62	+	0.33
	Tay	0.20		0.18	0.33		0.40
Cost *Non-Visitor*Non-Resident	All	-0.02	***	0.00	-		-
	Clyde	-0.03	***	0.00	-		-
	Forth	-0.02	***	0.00	-		-
	Tay	-0.01		0.00	-		-
ASC	All	-2.87	***	0.30	3.66	***	0.30
	Clyde	-2.45	***	0.53	3.87	***	0.52
	Forth	-3.02	***	0.56	3.96	***	0.60
	Tay	-2.91		0.43	2.82	***	0.40
Log-likelihood	All	-2851.58					
	Clyde	-901.68					
	Forth	-959.03					
	Tay	-954.98					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.26					
	Clyde	0.27					
	Forth	0.24					
	Tay	0.25					
AIC	All	5759.15					
	Clyde	1859.36					
	Forth	1974.05					
	Tay	1965.97					
BIC	All	5931.92					
	Clyde	2001.03					
	Forth	2116.15					
	Tay	2108.21					

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

4.3.4. Comparative analysis of willingness to pay for ecosystem services

The following section presents a comparative analysis of the welfare estimates which describe respondents' annual average WTP for a unitary change in a single attribute. This marginal WTP can be computed as the ratio between the respective ES attribute coefficient and the fixed cost coefficient. The confidence intervals (CI) were calculated with the Krinsky and Robb (1986) parametric bootstrap procedure using 1,000 replications of the unconditional parameter estimates.

The marginal WTP estimates presented in table 4-7 were estimated for all attributes using the MNL and RPL models without interactions since the ratio between the ES attribute coefficients and the cost coefficients has a more straightforward interpretation. Table 4-8 presents the same models but contrasts the annual average WTP estimated for each case study. Finally, table 4-9 collect the estimates of the RPL using attribute interactions (see tables 4-4 to 4-6) for computing the annual average WTP for the user types.

Overall, all estuarine ES are positively and significantly valued with values differing among ES, across catchment areas and between user types. As expected, the CI estimated for the pooled dataset with larger samples are narrower than the ones estimated for smaller samples, i.e. the estimates are more precise.

The ranking (in terms of marginal WTP) of ES obtained from the pooled sample (table 4-7), for each case study (table 4-8) and particular user types (table 4-9), is consistent with the one found for the MNL and RPL models (table 4-2). The outputs of the simple RPL model (using the pooled dataset) are plotted in figure 4-1. We found that flood control is the most highly valued estuarine ES, followed closely by biodiversity. The average annual WTP for recreation is significantly smaller (by at least a factor of two) when compared to both, flood control and biodiversity (see rows 1 to 6 in annex 11). Interestingly, these results are consistent with those presented in the previous chapter where respondents are explicitly asked to rank the three estuarine ES in order of importance (see table 3-6). Moreover, this ranking of ES by marginal WTP is consistent with that observed in Birol et al. (2009b), who valued comparable ES in a wetland ecosystem.

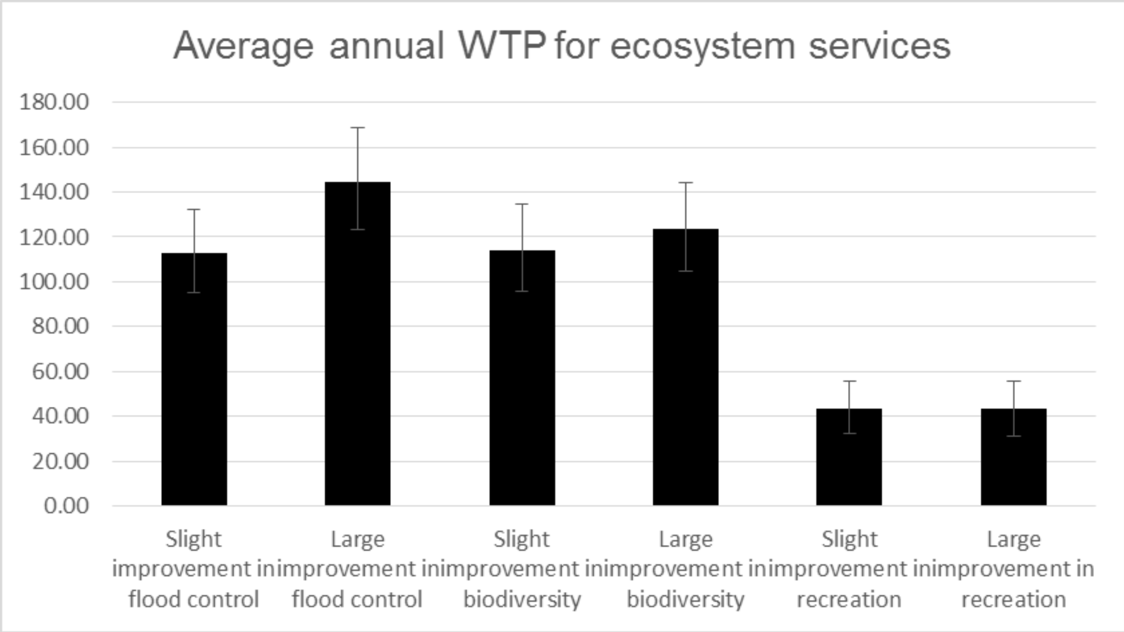


Figure 4-1 WTP estimates and 95% CI for ES improvements (GBP/year)

There are a few exceptions to this ranking of estuarine ES. For example, the outputs of the RPL model estimated with the pooled and the Clyde dataset indicate that when accounting for random preference heterogeneity lesser levels of biodiversity improvements (B1) are valued more than small improvements in flood control (F1) (see estimates plots in annex 12). Furthermore, when considering the user-specific WTP not estimates, we found that respondents who are 'residents' and 'both residents and visitors' valued biodiversity over flood control. These findings are in agreement with the previous discussion about preference coefficients which indicates that residents significantly valued biodiversity the most, regardless of whether or not they have visited the area for outdoor recreation or not (see estimates plots in annex 13 and rows 7 to 10 in annex 11). The results also indicate that visitors attach a significantly higher value to improvements in flood control, regardless if they reside in the area or not (see estimates plots in annex 13 and rows 11 to 14 in annex 11).

It is possible to observe from the three tables presenting WTP estimates (table 4-7 to table 4-9) that annual average WTP estimates for large improvements are generally associated with higher ES values. However, the average WTP does not increase with the level of improvements in biodiversity (B2) for the Clyde model, recreation (R2) in the Forth model and biodiversity (B2) for the resident sample. Therefore in these models, the increase in respondent utility is not higher for the large improvements as opposed to the more moderate improvements.

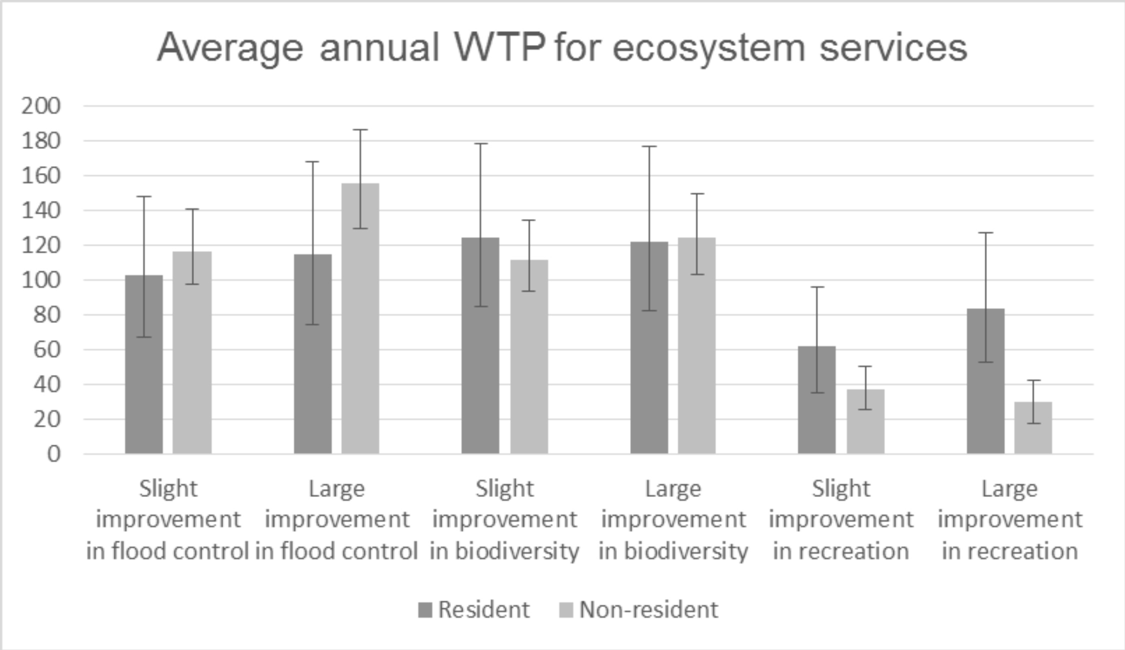
Table 4-8 (or annex 12) reveals a clear pattern for the improvements in flood control and biodiversity, which indicates that respondents are willing to spend the most for improving these estuarine ES inside the Tay catchment area, and the least for improving their levels in the Clyde catchment area. However, these differences of WTP estimates among areas is not significant for most of the cases (see rows 15 to 26 in annex 11). Additionally, it can be noted that the Clyde area is consistently ranked in the second position for improvements in the recreational services. The Tay catchment area is less populated and has a good environmental status (Tayside Biodiversity Partnership, n.d.). Thus people might target those ES which maintains or improves the current situation. Contrarily to more urban areas with lower environmental quality, such as the Clyde catchment (Clyde River Foundation, 2009), where respondents rather focus their expenditure on recreational services. This finding is consistent with the literature suggesting that the environmental baseline levels and the subjectively perceived *status quo* influence respondent’s WTP for environmental policies (Artell et al., 2013; Domínguez-Torreiro and Soliño, 2011; Kataria et al., 2012; Marsh et al., 2011; Meyerhoff and Liebe, 2009; Whitehead, 2013).

Regarding the ‘user type’ analysis, we found no recognisable patterns of WTP (see table 4-9 or annex 13) as no particular user type was found to be consistently associated with the highest values. Nonetheless, when we extend the analysis to contrast the user types with their respective non-use type (mirror variable), we found that some patterns emerge. Figure 4-2-a illustrates that the resident category is not always associated with significantly higher annual average WTP estimates when compared to the non-resident group. The t-test results shown in rows 27 to 32 of annex 11 support this finding. However, the visitor category presented a consistently and significantly higher average WTP when compared to the non-visitor category (see figure 4-2-b and rows 33 to 38 in annex 11). In other words, visiting the area matters for ES restoration and impacts the annual average WTP of respondents in a positive way. The present findings seem to be consistent with Sale et al. (2009) who use CV and find that the respondent status (e.g.

visitor vs non-visitor) is a significant predictor of the WTP for increasing the environmental condition in the Kromme estuary in South Africa.

Finally, when linking the WTP results (table 4-8 and table 4-9) with the ASC analysis presented in table 4-3, we can conclude that only occasionally the strength with which people preferred ES improvements was reflected directly in the magnitude of their WTP. For instance, among all case studies, the lowest preference for ES improvements (i.e. least negative *group-specific ASC*) and the smallest WTP estimate was associated with the Clyde catchment area. However, the preference for environmental improvement was not reflected directly in the magnitude of visitor's WTP estimates. Among all user types, visitors exhibit the strongest preference for ES improvements (i.e. most negative *group-specific ASC*), but only *assigned* the highest WTP for one ES, which is flood control improvements.

a)



b)

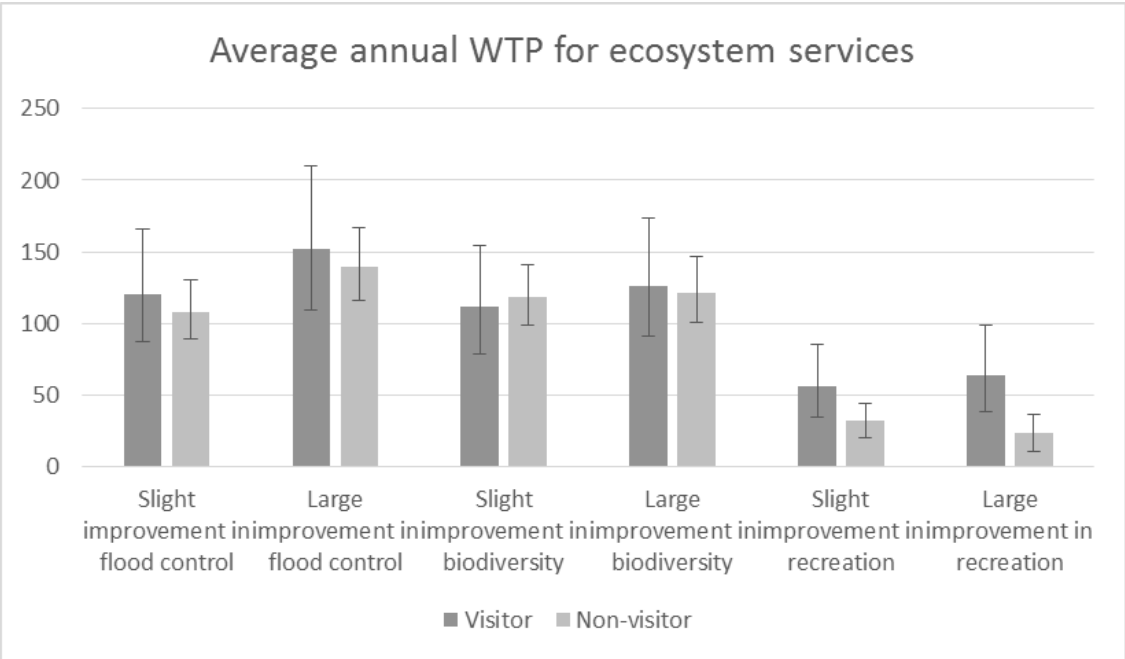


Figure 4-2 Direct-users vs indirect-users WTP estimates and 95% CI for ES improvements (GBP/year)

The top graph a) contrast the WTP estimates for the respondents who have visited the study area versus the ones who have not. The second graph b) compares the WTP estimates of residents against people residing outside the study area.

Table 4-7 WTP estimates for ES improvements

	MNL					RPL				
Attribute	WTP	C.I.				WTP	C.I.			
Flood control										
Slight improvement	113.54	(	98.61	129.84	)	112.89	(	95.28	132.05	)
Large improvement	141.43	(	125.06	160.99	)	144.42	(	123.43	168.77	)
Biodiversity										
Slight improvement	101.63	(	86.59	117.84	)	114.10	(	95.91	134.83	)
Large improvement	111.27	(	95.74	126.90	)	123.48	(	104.59	144.26	)
Recreation										
Slight improvement	37.85	(	25.58	49.27	)	43.68	(	32.47	55.77	)
Large improvement	40.17	(	29.73	51.58	)	43.24	(	31.22	55.86	)

Unit GBP. All models assumed a non-random cost coefficient and used 1000 Sobol draws for simulation. Confidence intervals use the Krinsky and Rob (1986) bootstrap method with 1000 draws. Parenthesis indicate the size of the confidence interval. WTP estimates were computed with the pooled dataset.

Table 4-8 Site-specific WTP estimates for ES improvements

	All					Clyde					Forth					Tay				
Attribute	WTP	C.I				WTP	C.I				WTP	C.I				WTP	C.I			
Flood control																				
Slight improvement	112.89	(	95.28	132.05	)	102.07	(	79.13	133.42	)	118.21	(	86.34	163.96	)	124.78	(	92.18	165.67	)
Large improvement	144.42	(	123.43	168.77	)	131.48	(	100.80	171.80	)	134.56	(	95.93	186.76	)	178.03	(	134.64	241.18	)
Biodiversity																				
Slight improvement	114.10	(	95.91	134.83	)	109.68	(	82.28	144.35	)	114.24	(	81.78	156.56	)	118.86	(	88.40	158.87	)
Large improvement	123.48	(	104.59	144.26	)	98.79	(	72.87	128.88	)	130.42	(	95.72	177.22	)	149.96	(	111.34	200.98	)
Recreation																				
Slight improvement	43.68	(	32.47	55.77	)	40.71	(	23.11	59.76	)	54.16	(	32.77	78.26	)	39.35	(	18.53	62.91	)
Large improvement	43.24	(	31.22	55.86	)	42.15	(	22.93	63.94	)	40.34	(	18.68	66.68	)	49.42	(	29.74	73.45	)

Unit GBP. All models assumed a non-random cost coefficient and used 1000 Sobol draws for simulation. Confidence intervals use the Krinsky and Rob (1986) bootstrap method with 1000 draws. Parenthesis indicate the size of the 95% confidence interval. WTP estimates were computed with the pooled as well as the site-specific datasets.

Table 4-9 User type specific WTP estimates for ES improvements

	Resident		Visitor		Resident & visitor	
Attribute	WTP	CI	WTP	CI	WTP	CI
Flood control						
Slight improvement	102.54	(67.42 147.94)	120.57	(87.01 166.19)	109.98	(72.75 170.24)
Large improvement	115.05	(74.71 167.65)	151.89	(109.45 210.00)	124.32	(79.74 191.99)
Biodiversity						
Slight improvement	124.08	(84.75 178.25)	111.87	(79.09 154.59)	133.05	(92.82 201.20)
Large improvement	122.22	(82.10 176.51)	126.55	(91.15 174.00)	138.32	(98.96 206.78)
Recreation						
Slight improvement	61.78	(35.12 96.14)	56.48	(34.72 85.73)	67.01	(38.52 109.50)
Large improvement	83.48	(52.59 127.26)	64.20	(38.73 98.97)	94.79	(59.55 148.43)

Unit GBP. All models assumed a non-random cost coefficient and used 1000 Sobol draws for simulation. Confidence intervals use the Krinsky and Rob (1986) bootstrap method with 1000 draws. Parenthesis indicate the size of the 95% confidence interval. WTP estimates were computed with the pooled dataset.

4.4. Conclusions and policy implications

This chapter uses a DCE to estimate the WTP for improvements in flood control, recreation and biodiversity resulting from implementing an ES restoration project in three catchment areas in Scotland. The present analysis contributes to the valuation literature by augmenting the heterogeneity analysis while accounting for the linkages of estuaries with other ecosystems which potentially impact on their capacity to provide benefits to society. Different modelling techniques were used, including the MNL, the simple RPL model and the interacted RPL model (ASC or attributes). These models were used to undertake a comparative analysis of the sources of preferences heterogeneity for estuarine ES. In particular, we explored how environmental preferences and WTP estimates vary across case studies with different baselines of ES provision, and among user types differing in the degree of direct use of the ES in question.

The findings of this chapter revealed a positive and significant WTP for improving estuarine ES provision, although differences in WTP estimates exist for all ES, across catchments and between user types. Recreation values were found to be lower on average than either flood control or biodiversity conservation. It has to be noted that this outcome does not necessarily suggest that respondents *assigned* higher values to all regulating services and lower values to all cultural services. First, our valuation analysis was limited to one ES per category. Second, it has been suggested that the magnitudes of the values of the cultural services significantly increase when studies expand the analysis from only including 'relational' values (e.g. visits to the area) to further explore the emotional and spiritual benefits (Chan et al., 2011; Stålhammar and Pedersen, 2017).

Furthermore, the results suggest that respondents from all over Scotland attach the highest WTP to the area with the highest current environmental quality (Tay area), and vice-versa. This finding could suggest that the rate on which environmental quality declines in an area might be more of a determinant for the WTP for ES improvements, than the baseline status of the environmental quality. However, this hypothesis needs to be tested formally in future studies.

Finally, our findings suggest a mismatch of priorities for different ES between residents and visitors. Results suggest that respondents who undertake visits to the area for developing outdoor recreation are on average more willing to fund a project which restores this and other estuarine ES. Since our results show that the effect of being a resident on WTP estimates is not as significant as being a visitor, we suggest that the proximity to the area is less important than the degree to which users engage with the studied area and make use of the recreational services.

Further analysis indicated that preference heterogeneity (associated with both the ASC and ES attributes) for estuarine ES restoration could be explained to a certain degree by the characteristics of the study region, as well as the degree on which respondents make direct use of the ES. Finally, our analysis reveals that stronger preferences for improvements in estuarine ES provision are not always translated into higher WTP. The previous finding is particularly relevant to acknowledge equity and social justice concerns in policy making. Environmental justice refers to an individual's capacity to mitigate risks in their own life (Broughel, 2014), and our findings suggest that some people might be deprived of enjoying better environmental quality against their will.

Overall, the findings of the analysis presented in this chapter might help policy makers and regulators to design contextualised environmental management policies. These findings have three main policy implications. Firstly, a multi-objective estuarine management policy targeting both flood control and biodiversity improvements is more likely to be accepted by Scottish citizens. Thus we suggest that catchment-base plans use natural flood management and develop green corridors alongside the rivers as these measures could be beneficial for both, flood control and biodiversity (Natural Scotland, 2013). These measures are also aligned with the Water Framework Directive (European Commission, 2015) and The Scottish Biodiversity Strategy (Scottish Executive, 2004) as they would improve the wildlife and the quality of the water environment.

Secondly, we found that respondents are more willing to fund restoration projects in areas with an already decent environmental status. Regions which are more capable of attracting funds could be used as a focus of attention to subsidise restoration projects happening in other 'less attractive' regions. Finally, since visiting the areas matters, choosing management policies which are compatible with the promotion of sustainable outdoor recreational visits could increase the willingness of people to pay for estuarine

ES improvements. Visiting such areas might boost the sense of ‘place attachment’, which in turn has been suggested to promote PEB in individuals (Halpenny, 2010; Ramkissoon et al., 2013a, 2012; Scannell and Gifford, 2010).

Although the DCE technique is useful for retrieving welfare estimates, it is also important to recognise its limitations when informing policy makers. DCE estimates are based on hypothetical scenarios and are applied in particular scenarios and contexts, thus potentially limiting their capacity of predicting individuals’ real behaviour and for extrapolating results into considerably different scenarios (Adamowicz et al., 1997; Hicks, 2002; Ready et al., 2005). This issue is of particular relevance when dealing with environmental attributes and levels that can be interpreted differently or are relatively unknown for respondents, which is the case of many coastal and marine ES (Jefferson et al., 2014; Jobstvogt et al., 2014a; Rose et al., 2008; Sandorf et al., 2017).

Other aspects of the DCE design should be taken into account when interpreting the results of environmental valuation studies, such as the type of *payment vehicle*. *Payment vehicle bias* might occur when respondents are unfamiliar with the use of tax levies (Morrison et al., 2000). The council tax is a plausible *payment vehicle* in this study as Scottish citizens are already paying it as a way of “buying” local public goods. For instance, Needham et al. (2018) indicated that local authorities are responsible for funding flood defence in Scotland.

It has to be noted that council taxes vary geographically in the UK and the proposed fixed increase in this study would result in different percentage increases across Scotland. Therefore, in order to avoid biases in the WTP estimations that could arise from regional variations on the council tax, it is critical to account for income differences when analysing respondent’s choices (as it has been done in this study). Although the effect the *payment vehicle* used was not tested formally, we followed literature recommendations and designed it within the institutional and cultural context in order to ameliorate this bias (Mitchell and Carson, 1989; Morrison et al., 2000). Moreover, recent empirical research has found no significant differences in both, marginal utility and choice consistency associated with the type of *payment vehicle* used in valuation studies (Gibson et al., 2016; Koetse, 2017).

On other regardss, even though the analysis used a representative sample, our study sample has some limitations. Firstly, funding restrictions limited our sample size and led to a reduction in the significance levels of the coefficients obtained in the subset datasets. Moreover, our study sample also presents spatial limitations with a large part of the people surveyed clustered around the main population settlements, i.e. the Central Belt in Scotland. This matter is not particularly relevant for the analysis of this chapter, but further studies can be developed to test if the spatial representativeness of the sample matters for generating more precise WTP estimates.

The present analysis has explored how the socioeconomic characteristics of the respondents can explain the heterogeneity of WTP estimates. Nonetheless, it is likely that some other exogenous factors influence ES values, such as the spatial distribution and the relative scarcity of natural resources in the catchment areas. There is a need for future environmental valuation research to account for spatial factors, thus the authors develop this analysis in the following chapter.

Chapter 5. Spatial context effect on preference heterogeneity

5.1. Introduction

Recent environmental valuation literature has revealed a spatial dimension of preference heterogeneity. Variations of mean WTP in the geographical space have been characterised by the presence of localised patches of higher and lower values (Johnston et al., 2015; Johnston and Ramachandran, 2014; Meyerhoff, 2013). However, to date, spatial applications of environmental valuation studies have not explored whether the distribution of hot (cold) spots of WTP estimates is particular to each environmental good or if instead, it follows similar patterns to other, comparable, environmental goods.

This chapter presents the empirical analysis addressing the *Specific objective 2* and answering the questions derived from it. As explained in chapter 1, the second empirical analysis aims to compare the geographical distribution of local clusters of WTP for improvements in estuarine ES. Chapter 2 has already explained why the local spatial context is expected to affect environmental preference heterogeneity. Thus this chapter is particularly interested in analysing whether the geographical patterns of preference heterogeneity are constant among ES and across study sites.

Generating further understanding of the effects of spatial context on preference heterogeneity could facilitate the design of more efficient policies which use spatially explicit designs. Moreover, finding general patterns regarding society's WTP distribution can reduce the complexity of the information that valuation practitioners provide to policy makers.

This chapter, together with the analysis in chapters 4 and 6 aims to generate information to guide policy makers and regulators in designing effective and contextualised environmental management policies. We utilised geocoded individual-specific WTP data derived from a RPL model in *WTP space*. Afterwards, the local Moran's I statistic was used to find statistically significant local clusters of WTP. Finally, the Multi-type Ripley's K function is used to contrast the spatial patterns of local clusters of WTP among estuarine ES and across case study estuaries.

Research on spatial preference heterogeneity has tended to focus on environmental goods (Czajkowski et al., 2016; Johnston et al., 2015; Meyerhoff, 2013), rather than ES. Therefore our analysis contributes to the environmental valuation body of knowledge in two main ways. First, by exploring the existence of significant spatial heterogeneity in environmental preferences for estuarine ES improvements. Secondly, we compare the distribution of local clusters of WTP estimates for different estuarine ES and among different regions.

The rest of this chapter is organised as follows. Section 5.2 describes the CM framework; the statistics used to find local clusters of WTP and the summary functions used to contrast them. Afterwards, we present and discuss the results of the econometric models, as well as the comparative analysis of local clusters of WTP estimates (see section 5.3). Finally, section 5.4 presents a synthesis of the main findings and discuss their policy implications.

5.2. Empirical analysis

The choice dataset used in this chapter was obtained from a DCE conducted in Scotland in 2016 to estimate society's WTP for improving flood control, recreation and biodiversity within the Clyde, Forth and Tay catchment areas. Details regarding the DCE design can be found in chapter 3. As it was explained in table 3-7, the analysis of this chapter uses the choices of a sample of 571 individuals, which was obtained after correcting for protest respondents (1.80%) and deleting individuals without postcode and income information. T-tests show that this sample is representative of the Scottish population in most of the available statistics, except age (see annex 14 for the summary of the sample descriptive statistics).

After testing for several specification forms, we defined the utility as a linear function of dummy coded attributes and the ASC, as it fits our data better. Table 5-1 describes the coding used in the model output tables. All models were coded and estimated in R software (version 3.3.2) using the pooled dataset, as well as with the site-specific datasets.

Table 5-1 Explanation of variable abbreviations and coding

Variable	Explanation
ASC	Constant term (0 = Option1: NO new policy, 1 = Option 2 or 3)
F1	Change in flood control from "increase in flood risk" to "slight reduction in flood risk" (1 = yes, 0 = no)
F2	Change in flood control from "increase in flood risk" to "large reduction in flood risk" (1 = yes, 0 = no)
B1	Change in biodiversity from "decrease in biodiversity" to "slight increase in biodiversity" (1 = yes, 0 = no)
B2	Change in biodiversity from "decrease in biodiversity" to "large increase in biodiversity" (1 = yes, 0 = no)
R1	Change in recreation from "decrease in recreation" to "slight increase in recreation" (1 = yes, 0 = no)
R2	Change in recreation from "decrease in recreation" to "large increase in recreation" (1 = yes, 0 = no)
Cost	Additional council tax payment
Resident	Whether respondent resides in the catchment area (1 = yes, 0 = no)
Visitor	Whether respondent visited the area for outdoor recreational activities in the last 12 months (1 = yes, 0 = no)
Female	Respondent's gender (1 = Female, 0 = Male)
Age	Respondent's age is above the average (1 = yes, 0 = no)
Graduate	Whether respondent has undergraduate and/or postgraduate education (1 = yes, 0 = no)
Income	Respondent's income is above the average for the sample (1 = yes, 0 = no)

5.2.1. Choice modelling

The utility of an alternative i for respondent n in the choice occasion t is given by:

$$U_{int} = V_{int} + \varepsilon_{int} \tag{5-1}$$

$$V_{int} = f(\beta, x_{int}, z_n) \tag{5-2}$$

where β_n is a vector of utility weights of respondent n, x_{int} is a vector of attributes of alternative i in choice occasion t, z_n is a vector of measured attributes of respondent n and $\varepsilon_{i,n,t}$ is a random term which is assumed to be distributed IID extreme value.

Nonetheless, since the model is specified in preference space the utility of an alternative i for respondent n as separable in cost c, and non-price attributes x becomes:

$$U_{int} = -\alpha_n \cdot c_{int} + \beta_n' \cdot x_{int} + \varepsilon_{int} \tag{5-3}$$

where α_n and β_n vary randomly over respondents and ε_{int} is again assumed to be distributed IID extreme value with a variance given by $\mu_n^2(\Pi^2/6)$, where μ_n is an individual-specific scale parameter. Train and Weeks (2005) showed that equation 5-3 can divided by μ_n without affecting behaviour and results giving a new error term which is IID extreme value distributed with a variance equal to $\Pi^2/6$. This utility is specified in preference space and is written as:

$$U_{int} = -\lambda_n \cdot c_{int} + c_n' \cdot x_{int} + \varepsilon_{int} \tag{5-4}$$

where $\lambda_n = \alpha_n/\mu_n$ and $c_n = \beta_n/\mu_n$. Considering the fact that the WTP for the attributes is given by the ratio $\gamma_n = c_n/\lambda_n$. Equation 5-4 can be rewritten now in *WTP space* as:

$$U_{int} = \lambda_n[c_{int} + \gamma_n' \cdot x_{int}] + \varepsilon_{int} \tag{5-5}$$

The model estimated in this chapter is the best-fitting model from chapter 4 (RPL interacted 1) specified in *WTP space*. The re-parameterised model in *WTP space* directly provides the marginal value for each attribute level and permit us to account for the taste heterogeneity in the price coefficient (Sonnier et al., 2007; Train and Weeks, 2005b).

The V_{int} component in *WTP space* RPL model is written as:

$$\begin{aligned} V_{int} = {} & ASC_{SQ_{int}} + \sigma_{ASC_{SQ_{int}}} \cdot \xi_{1,n} \cdot ASC_{SQ_{int}} + ASC_{resident} \cdot \mathrm{Z}_{resident,n} + \\ & ASC_{visitor} \cdot \mathrm{Z}_{visitor,n} + ASC_{female} \cdot \mathrm{Z}_{female,n} + ASC_{age} \cdot \mathrm{Z}_{age,n} + \\ & ASC_{graduate} \cdot \mathrm{Z}_{graduate,n} + ASC_{income} \cdot \mathrm{Z}_{income,n} - \beta_{Cost}\,\big[-Cost_{int} + \\ & \beta_{\mathrm{F1}} \cdot \mathrm{F1} + \sigma_{\mathrm{F1}} \cdot \xi_{1,n} \cdot \mathrm{F1} + \beta_{\mathrm{F2}} \cdot \mathrm{F2} + \sigma_{\mathrm{F2}} \cdot \xi_{1,n} \cdot \mathrm{F2} + \beta_{\mathrm{B1}} \cdot \mathrm{B1} + \sigma_{\mathrm{B1}} \cdot \xi_{1,n} \cdot \\ & \mathrm{B1} + \beta_{\mathrm{B2}} \cdot \mathrm{B2} + \sigma_{\mathrm{B2}} \cdot \xi_{1,n} \cdot \mathrm{B2} + \beta_{\mathrm{R1}} \cdot \mathrm{R2} + \sigma_{\mathrm{R1}} \cdot \xi_{1,n} \cdot \mathrm{R1} + \beta_{\mathrm{R2}} \cdot \mathrm{R2} + \\ & \sigma_{\mathrm{R2}} \cdot \xi_{1,n} \cdot \mathrm{R2}\big] \end{aligned} \tag{5-6}$$

where ASC_{SQ} is an alternative specific constant (which equals zero if respondents chose the *status quo* and one if they did not), with Z_n being a vector of measured sociodemographic variables of respondent n (in interaction with the ASC), and the attributes are used as described in table 5-1.

Our RPL model assumed the cost parameter to be constant (fixed) across respondents with randomly distributed parameters for the ES attributes and the ASC_{SQ}. In this specification, $\xi_{1,n}$ is a random variable that follows a standard normal distribution across individual respondents but is held constant across choices for the same respondent n. The attributes thus follow a Normal distribution across respondents, with mean β and standard deviation σ. The density for β is denoted as $f(\beta|\theta)$, where θ are the parameters of the distribution. This model accounts for both a stochastic and a systematic component of heterogeneity by allowing preference deviation around the mean population parameter for attributes, and by including ASC to interact with specific socioeconomic characteristics.

Let $L(y_n|\beta)$ give the likelihood of the observed sequence of T_n choices of respondent n, P_n is the product of discrete choice probabilities depending on the model assumptions. The probability of the observed sequence of choices conditional on knowing β_n is given by:

$$P_n(\beta_n) = \Pr(y_n^t|\cdot) = \prod_{t=1}^{Tn} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{ijt}}} \tag{5-7}$$

The unconditional probabilities are the integrals of the standard logit probabilities over a density of parameters. The choice probability for person n can be expressed by:

$$P_{int} = \Pr(y_n^t|\cdot) = \int P_n(\beta)\, f(\beta|\theta) d\beta \tag{5-8}$$

In our application $f(\beta)$ is specified to be continuously distributed $\beta \sim f(\beta|\theta)$ with the vector of parameters β and a covariance matrix Ω and $\beta \sim N(0|1)$ following a normal distribution. The choice probability for person n is thus given by:

$$P_{int} = \Pr(y_n^t|\cdot) = \int_\beta \prod_{t=1}^{Tn} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{ijt}}}\; f(\beta|\theta) d\beta \tag{5-9}$$

Considering that our model is in *WTP space*, the equation 5-9 is rewritten as:

$$P_{int} = \Pr(y_n^t|\cdot) = \int_{\alpha_n} \int_{\beta_{cost}^n} \prod_{t=1}^{Tn} \sum_{i=1}^{I} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{ijt}}}\; f(\alpha_n, \beta_{cost}^n|\theta) d\alpha_n d\beta_{cost}^n \tag{5-10}$$

Finally, the model is estimated by maximum likelihood. The log-likelihood function for the model is given by $LL(\theta) = \sum_{n=1}^{N} \ln P_{int}$. This expression cannot be solved analytically and simulation-based estimation of the model is used to evaluate P_n at a large number of draws from β, in our case 1,000 Sobol draws. Similarly to the analysis in chapter 4, the fixed cost parameter assumption was made to avoid convergence issues and to facilitate the implicit prices (i.e. WTP) calculation (Revelt and Train, 1998; Wielgus et al., 2009).

The simulated log likelihood of the RPL model is given by:

$$LL(\theta) = \sum_{n=1}^{N} \ln \left[\frac{1}{R}\sum_{r=1}^{R} P_n(\beta^r)\right] \quad 5\text{-}11$$

Alternatively,

$$LL(\theta) = \sum_{n=1}^{N} \ln \int_{\beta} P_n f(\beta|\theta) d\beta \quad 5\text{-}12$$

Which is rewritten in *WTP space* as:

$$LL(\theta) = \sum_{n=1}^{N} \ln \int_{\alpha_n} \int_{\beta_{cost}^n} P_n \; f(\alpha_n, \beta_{cost}^n|\theta) d\alpha_n d\beta_{cost}^n, \quad 5\text{-}13$$

where P_n is defined as in equation 5-7. In a RPL model, the parameters of β distribution (θ) are estimated, rather than a vector of β point values as is done when estimating a MNL. The RPL model also provides individual-specific posterior estimators of the parameter vector as well as the individual-specific conditional distributions of the random parameters (Huber & Train, 2001).

If $L(Y_n|\beta)$ is the probability of observing the specific value for β, then (Train, 2003) the probability of observing an specific value of β for T parameters, given respondent n choices is defined as:

$$T(\beta|Y_n) = \frac{L(Y_n|\beta)\, f(\beta|\theta)}{\int_{\beta} L(Y_n|\beta)\, f(\beta|\theta) d\beta} \quad 5\text{-}14$$

We followed the approach in Hess (2007) to calculate individual-specific draws from the random distributions conditioned on the observed sequence of choices for each respondent. This approach proved to be useful as it reduces problems related to biased trade-offs when calculating the individual-specific ratios.

In this way, we replaced the continuous formulation by a discrete approximation using summation over a high number of draws. The mean for the conditional distribution for respondent n is given by:

$$\widehat{\beta_n} = \frac{\sum_{r=1}^{R}[L(Y_n|\beta_r)]}{\sum_{r=1}^{R} L(Y_n|\beta_r)} \qquad \text{5-15}$$

where β_r with $r = 1, \ldots, R$ are independent multidimensional draws with equal weight from $f(\beta|\theta)$ at the estimated values for θ. Once $\widehat{\beta_n}$ are known, it is possible to calculate a single value for each trade-off per respondent as well as the individual-specific distributional statistics. The specification in *WTP space* of our model allows us to directly obtain from the posterior analysis the individual-specific marginal WTP estimates for each attribute, as well as their distribution. This model was applied to the pooled choice dataset as well as to each catchment-specific datasets (see table 5-2). As we used three ES attributes with two improved levels, we obtained six conditional WTP estimates per respondent and for each RPL model.

Data on household postcode information (postcode centroid) was used to geocode the location of each respondent across Scotland and was linked to their individual-specific conditional mean WTP estimates as explained above. The postcode unit represents a relatively precise measure of a respondent's residential location since each postcode in the UK covers an average of only 15 properties (Ordnance Survey, 2018).

5.2.2. Spatial analysis of willingness to pay for ecosystem services

In contrast to Campbell et al. (2009, 2008) who developed a spatial autocorrelation analysis in a polygon format (by averaging administrative areas), we used a point analysis that accounted for the irregular distribution of the sample in space by using k-nearest neighbour weight matrices (see equation 5-19). This was done for several reasons. First, we collected a non-homogenously distributed sample data in space, therefore averaging this data would result in imbalanced or biased estimates. Second, as the 'modifiable areal unit problem' suggest averaging counties or states can influence the strength of measured spatial autocorrelation (Meyerhoff, 2013; Openshaw, 1983). Finally, and most importantly, averaging data points inside an administrative area imposes a general trend within the limits of the area and might obscure the presence of 'patchy' patterns of marginal WTP for estuarine ES improvements in Scotland.

The geo-referenced individual-specific mean WTP estimates of all datasets (summarised in table 5-3) were used to test for the presence of spatial autocorrelation. We first calculated global measures of autocorrelation, as it has been suggested that the presence of significant global autocorrelation could increase the probability of incorrectly identifying local spatial autocorrelation (Ord and Getis, 2001).

We tested for global spatial autocorrelation with the Moran's I statistic (Moran, 1950) defined as:

$$I = \frac{n\sum_{i=1}^{n}\sum_{j=1}^{n}\omega_{ij}(WTP_i - \overline{WTP})(WTP_j - \overline{WTP})}{S_0\sum_{j=1}^{i=n}(WTP_i - \overline{WTP})^2}, i \neq j \qquad \text{5-16}$$

where ω_{ij} is the weight between observation i and j, and S_0 is the sum of ω_{ij}'s:

$$S_0 = \sum_{i=1}^{n}\sum_{j=1}^{n}\omega_{ij} \qquad \text{5-17}$$

The underlying assumption of global Moran's I test is that the spatial process promoting the observed estimates of WTP for ES restoration is a random chance. Rejecting the null hypothesis would suggest the existence of spatial autocorrelation of WTP estimate values. The value of Moran's I ranges between +1 and -1, with positive values indicating that WTP estimates are globally clustered (or positive autocorrelation) and negative values indicating they are globally dispersed (or negative autocorrelation). Values of I ~ 0 indicate that WTP estimate values are distributed randomly in space.

Since the Moran's I statistic consist of the summation of individual cross products, the local indicator of spatial autocorrelation (LISA) can be used to evaluate clustering in those individual units by estimating the local Moran's I index for each spatial unit and evaluating the statistical significance for each I_i. Thus we subsequently used the LISA statistics to test if spatial autocorrelation is more likely to occur within subsets of datasets (Anselin, 1995). In other words, the LISA statistics help to identify for each observation of WTP whether significant local clustering is occurring.

We applied the local Moran's I_i defined as by Getis (2010):

$$I_i = \frac{WTP_i - \overline{WTP}}{\left((1/n)\sum_{j=1}^{i=n}(WTP_i - \overline{WTP})^2\right)}\sum_{j=1}^{n}\omega_{ij}(WTP_i - WTP_j), i \neq j \qquad \text{5-18}$$

The local Moran's I_i values for the pooled and the site-specific datasets were classified into five categories (see table 5-5), including insignificant clusters as well as four types of significant clusters. The first two types of significant clusters are hotspots (HH) and coldspots (LL). The former represents respondents with high values having neighbours with high values, whereas the latter refers to respondents with low values surrounded by neighbours with low values. The remaining two categories are respondents with high values which have neighbours with low values (HL) and respondents with low values which have neighbours with high values (LH).

The local Moran's I_i estimations used a k-nearest neighbour weight matrix definition. Let the centroid distances from each spatial unit i to all units $j \neq i$ be ranked as follows: $d_{ij(1)} \leq d_{ij(2)} \leq d_{ij(n-1)}$. Then for each $k = 1, \dots, n-1$, the set $N_k(i) = \{j(1), j(2), \dots, j(k)\}$ contains the k closest units to i. For each given k, the k-nearest neighbour weight matrix (W) then has spatial weights defined as follow:

$$\omega_{ij} = \begin{cases} 1, j \in N_k(i) \\ 0, otherwise \end{cases} \qquad \text{5-19}$$

The use of k-nearest neighbours for spatial weights is more appropriate when dealing with non-homogenously distributed sample points because it defines the neighbourhood around any given point as the closest k neighbouring points, regardless of distance. We conducted a sensitivity analysis and tested for different numbers of neighbours used to define the spatial weight matrix (k values from 8 to 100) when estimating both, global and local Moran's I statistics. Global spatial autocorrelation was maximised at different k values, however, results are consistent for different values of k (see annex 15). Furthermore, differences in the proportion of significant local clusters[15] were very small and for the purposes of our analysis were inconsequential (see annex 16). Thus based on the above findings, we follow the suggestion by Duda et al. (2001) of using $k = \sqrt{n}$ and report the results for $k = 23$ (closest twenty-three data points) in the following analysis.

The *spedep* package in R (Bivand and Piras, 2015) was used to estimate the spatial weights matrix, as well as to conduct both global and local measures of spatial autocorrelation.

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[15] Estimated as the ratio of the number of local clusters within a category and the total number of geocoded data (e.g. number of HH/number of respondents).

5.2.3. Summary functions for comparing spatial point patterns

The local clusters identified when analysing site-specific environmental preferences (site-specific datasets) were used to proceed with the comparison of the spatial patterns among estuarine ES and across the three catchment areas. We used a multi-type point pattern for HH and LL individuals treated as a single pattern of n points and marked by both the study case (three types) and the ES they refer to (three types). A marked point pattern is explained as an unordered set $y = \{(x_1, m_1), \dots (x_n, m_n)\}$, $x_1 \in W, m_1 \in M$, where x_i are the locations and m_i are the marks which allow grouping the points into types.

A preliminary visual examination of hot (cold) spots pattern plots was first developed to identify commonalities in the spatial trends among ES and case studies (see figure 5-2). Since it is difficult to draw general conclusions from the visual comparison of plots (Long and Robertson, 2018), we subsequently used the Multi-type Ripley's K function (cross-type) to test whether or not there is clustering between all pairs of types (Ripley, 1981).

Let X_j denote the sub-patterns of points type j, with intensity (density of points) λ_j. Then the bivariate K function for any pair of types i and j:

$$K_{ij}(r) = 1/\lambda_j E \tag{5-20}$$

where E is the expected number of points of type j within a distance r of a typical point of type i in the process X_i. The *spatstat* package in R was used to compute the estimator K_{ij} of the Ripley K-cross function. It assumes that X is a realisation of a stationary (spatially homogeneous) random spatial point process in the plane which is typically modelled as a Poisson point process, observed through a bounded window (Baddeley et al., 2015).

For graphing purposes, we used a variance stabilising transformation of the K-cross function that derives into the L-cross function defined by (Besag, 1977) as:

$$L_{ij}(r) = \sqrt{\frac{K_{ij}(r)}{\pi}} \tag{5-21}$$

Both summary functions K_{ij} and L_{ij} are called ‘cross-type’, ‘bivariate’ or ‘i-to-j’ when $i \neq j$. If X_i and X_j point processes are probabilistically independent then $K_{ij}(r) = \pi r^2$ and $L_{ij}(r) = r$, regardless of the pattern of either type of point (Ripley, 1981).

The first step in the analysis is focused on comparing the observed pattern of HH (or LL) to a Complete Spatial Randomness and Independence (CSRI) process (see first two rows in figure 5-3 and figure 5-4). Plotting r vs $L_{ij}(r)$ provides a convenient reference line at zero. Values of $L_{ij}(r) < r$ indicate inhibition between two types of points, whereas values of $L_{ij}(r) > r$ indicate more clustering than expected under CSRI.

Following Ripley (1977), we extended the exploratory analysis of HH (or LL) point pattern to include a Monte Carlo test of goodness-of-fit to the homogeneous Poisson process. In the last two plots of figure 5-3 and figure 5-4, the L_{ij} function was plotted together with a simulation of envelopes (1000 simulations), where each simulation is generated by the homogeneous Poisson point process with intensities estimated from the HH (or LL data). The envelopes serve as the critical limits for a Monte Carlo test of the null hypothesis of a random Poisson point process. If the observed L-cross function is outside the simulation envelope, it shows clustering between X_i and X_j. Clustering between a pair of types of points occurs when the events of each type are closer to each other than expected under the assumption that the two processes are independent.

The summary functions K_{ij} and L_{ij} allow us to develop a pair-wise analysis of all the possible combinations of the survey point pattern types defined inside our marks (‘ES-types’ and ‘survey-types’). For instance, they allow us to understand the interaction between the point pattern HH_{Clyde} and HH_{Tay} if focusing on the ‘survey-types’ marks, or to explore if the ‘ES-types’ point patterns of HH_{flood} and $HH_{biodiversity}$ are clustered together.

5.3. Empirical results and discussion

5.3.1. Random parameter logit model in willingness to pay space

Table 5-2 presents the results of the RPL model estimated in *WTP space*. It combines the results of the choice models derived from the (i) pooled dataset, as well as the site-specific choices for the (ii) Clyde, (iii) Forth and (iv) Tay estuaries. Similarly to the previous chapter, the site-specific estimates are stacked for easing models comparison. The respective model from which each estimate is derived is indicated in the column 'Dataset'. After testing for several forms to include attributes into the utility function, we selected a dummy coded specification for all non-monetary attributes.

Table 5-2 RPL estimates for ES improvement in WTP space

		RPL interacted					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1	All	111.30	***	6.84	42.74	***	8.76
	Clyde	100.16	***	9.61	34.82	**	12.94
	Forth	114.56	***	12.86	49.99	***	14.65
	Tay	118.99	***	13.62	44.65	*	18.02
F2	All	141.30	***	8.48	78.66	***	8.45
	Clyde	125.74	***	12.03	69.71	***	11.86
	Forth	129.21	***	15.03	88.08	***	16.66
	Tay	170.70	***	17.90	70.61	***	16.51
B1	All	114.00	***	7.08	0.69		21.22
	Clyde	105.45	***	10.51	1.75		31.77
	Forth	113.65	***	13.24	41.30	**	15.70
	Tay	117.45	***	13.59	0.80		21.92
B2	All	122.11	***	7.72	54.68	***	7.80
	Clyde	94.89	***	10.46	37.33	*	12.07
	Forth	128.71	***	14.42	58.94	***	15.06
	Tay	146.17	***	16.26	63.97	***	15.09
R1	All	42.21	***	5.04	2.80		14.04
	Clyde	38.78	***	7.40	0.27		21.63
	Forth	51.63	***	9.54	1.45		17.21
	Tay	36.91	***	9.68	6.62		44.25
R2	All	42.60	***	5.34	42.59	***	7.98
	Clyde	41.89	***	8.24	38.18	**	12.71
	Forth	38.50	***	9.91	54.72	***	13.59
	Tay	48.54	***	9.66	15.39		66.02
ASC	All	-0.08		0.52	3.11	***	0.25
	Clyde	0.81		0.93	3.40	***	0.49

Attribute	Dataset	RPL interacted					
		Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Forth	0.55		0.94	3.03	***	0.45
	Tay	-1.70	*	0.93	2.67	***	0.42
*ASC*resident*	All	0.39		0.43	-	-	-
	Clyde	0.38		0.85	-	-	-
	Forth	0.25		0.73	-	-	-
	Tay	-0.50		0.99	-	-	-
*ASC*visitor*	All	-1.05	**	0.41	-	-	-
	Clyde	-0.84		0.84	-	-	-
	Forth	-1.51	*	0.75	-	-	-
	Tay	-1.10		0.63	-	-	-
*ASC*female*	All	-0.73	*	0.38	-	-	-
	Clyde	-0.67		0.70	-	-	-
	Forth	-1.08		0.66	-	-	-
	Tay	-0.01		0.67	-	-	-
*ASC*age*	All	-1.05	**	0.38	-	-	-
	Clyde	-1.51	+	0.75	-	-	-
	Forth	-1.43	*	0.64	-	-	-
	Tay	-0.22		0.64	-	-	-
*ASC*graduate*	All	0.21		0.38	-	-	-
	Clyde	0.42		0.74	-	-	-
	Forth	-0.73		0.71	-	-	-
	Tay	0.99		0.63	-	-	-
*ASC*income*	All	-0.03	+	0.01	-	-	-
	Clyde	-0.06	*	0.03	-	-	-
	Forth	0.01		0.03	-	-	-
	Tay	0.00		0.02	-	-	-
Log-likelihood	All	-2662.65					
	Clyde	-870.54					
	Forth	-903.17					
	Tay	-864.69					
Observations	All	3426.00					
	Clyde	1128.00					
	Forth	1164.00					
	Tay	1134.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.28					
	Forth	0.28					
	Tay	0.29					
AIC	All	5367.29					
	Clyde	1783.09					
	Forth	1848.34					
	Tay	1771.39					

		RPL interacted			
Attribute	Dataset	Coeff. (Mean)	S.E.	Coeff. (S.D.)	S.E.
BIC	All	5496.21			
	Clyde	1888.68			
	Forth	1954.59			
	Tay	1877.09			

Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.
Source: Scottish estuarine management Choice Experiment, 2016.

With ρ^2 values between 0.28 and 0.29, the four RPL models applied to the choices for improvements in estuarine ES have relatively high explanatory power, as per Hensher and Johnson (1981), who indicate that models with ρ^2 values between the range of 0.2 and 0.4 in ρ^2 values could be considered good fits. Instead of presenting the attribute preference coefficients, table 5-2 shows the estimates of marginal prices WTP (mean and standard deviation) derived directly from the estimations in *WTP space*. As it can be seen, there is positive and significant mean WTP for all improvements in estuarine ES. The significantly negative cost coefficient shows respondent’s preference for management options with lower cost (all other attributes remaining equal).

The standard deviations reveal significant unobserved heterogeneity across all attribute levels, except for slight improvements in biodiversity (B1) and recreation (R1). Smaller WTP estimates (with smaller variability) are related to small estuarine ES level improvements and greater WTP estimates (with greater variability) for more substantial ES gains. In other regards, we found that the ASC does not always exhibit a negative sign, but in these cases, it fails to reach significance. The presence of a negative ASC suggests that on average respondents’ utility is impacted positively when moving away from the *status quo*.

As expected, the results regarding the user-specific ASC vary depending on the dataset analysed. However, it can be seen that for most of the cases the sign remains constant across datasets and that the significance of the coefficients is commonly reached for the larger sample (pooled dataset). Similarly to previous research (Birol et al., 2009; Börger and Hattam, 2017; Botzen et al., 2012), we found that for the pooled dataset the ASC of visitors, female and older people is negative and significant (at least at the 10% level), indicating their preference for moving away from the *status quo* and to develop projects delivering improvements in ES.

Finally, it can be noted that the higher WTP estimates are commonly associated with flood control and followed closely by biodiversity improvements. However, recreational changes in the catchment area present a value decrease of at least 55% when compared to the former and latter. This ranking of estuarine ES matches the one observed in simpler models estimated in the previous chapter and also accords with the findings in Birol et al. (2009b).

5.3.2. Individual-specific willingness to pay analysis

Table 5-3 displays a summary of the individual-specific WTP estimates for all estuarine ES improvements which were calculated using both, the pooled and site-specific datasets. These estimates can be interpreted as the mean, minimum and maximum value of the parameters of the subpopulation that would have made the same choices while facing the same choice situation. These individual-specific WTP estimates are used in the subsequent spatial analysis presented in section 5.3.3.

Results are in agreement with the findings of the previous chapter (see section 4.3) and reveal that higher disparities in WTP estimates (max-min) are found for the large flood control improvements (F2), whereas the substantially smaller distribution of WTP estimates relate to slight and large enhancements in recreational services (R1 and R2). This results might be explained by the variability of flood risk perception, which has been found to determine the WTP for flood control measures (Zhai et al., 2006).

Table 5-3 Individual-specific WTP estimates for ES improvements

Attribute	All		Clyde		Forth		Tay	
	Mean	Min-Max	Mean	Min-Max	Mean	Min-Max	Mean	Min-Max
Flood control								
Slight improvement	111.30	62.70 / 157.80	100.20	68.57 / 130.80	114.30	51.59 / 161.00	119.00	69.69 / 166.60
Large improvement	141.30	12.45 / 246.20	125.90	10.17 / 221.90	129.20	1.75 / 254.60	170.40	91.45 / 251.60
Biodiversity								
Slight improvement	114.00	113.80 / 114.20	105.50	105.20 / 105.70	113.70	67.95 / 152.60	117.50	117.40 / 117.90
Large improvement	122.00	54.97 / 204.10	95.03	55.83 / 149.00	128.90	62.31 / 204.60	146.10	51.58 / 228.10
Recreation								
Slight improvement	42.20	41.38 / 42.60	38.78	38.72 / 38.83	51.63	51.23 / 51.99	36.90	33.19 / 38.62
Large improvement	42.65	-10.35 / 100.90	41.96	-1.29 / 94.12	38.76	-33.90 / 121.50	48.51	42.19 / 55.68

Unit GBP/year.

5.3.3. Spatial autocorrelation of willingness to pay for ecosystem services

Figure 5-1 shows that the point density of the sample used in this analysis (n=571) is higher in the Central Belt of Scotland and the Aberdeen region which are the most densely populated areas of Scotland.

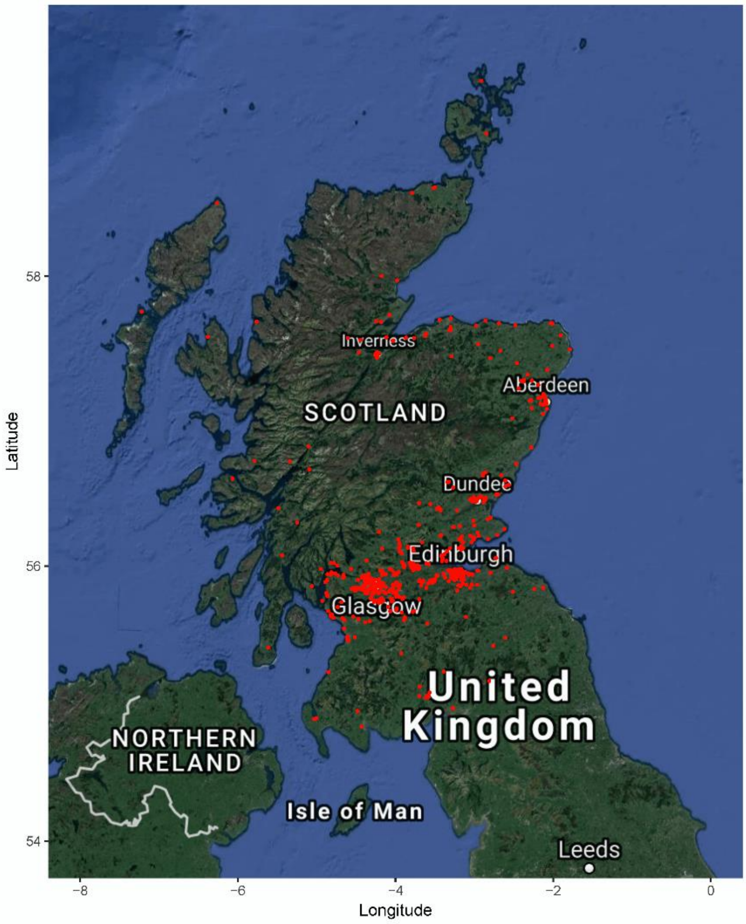

Figure 5-1 Geographical distribution of WTP sample points in Scotland

The geocoded individual-specific mean WTP data points obtained from the pooled and site-specific choice models were used to explore for spatial autocorrelation. We first tested for global spatial correlation using Moran's *I* statistic. Results are presented in table 5-4 and only indicate the presence of globally clustered WTP mean values for delivering slight improvements in flood control when analysing the pooled dataset. Johnston & Ramachandran (2014) argued that local spatial patterns of WTP estimates might exist even if global patterns of spatial significance are absent. Therefore we used the LISA and tested for locally spatially autocorrelated WTP estimates, in the pooled as well as the site-specific datasets.

In order to test for the robustness of our results at a different scale, we also applied the same analysis to a subset of both the pooled and site-specific datasets which only include the residents of the catchment areas (see annex 17 and 18). However, the robustness of this results is limited by the significant reduction of the sample size. Therefore, we proceed to develop the analysis with the full datasets and report the results from this analysis in this chapter.

We found local statistically significant clusters of WTP estimates for improvements in all estuarine ES attributes. The number of significant clusters was higher in the pooled dataset, as it is also the dataset with the largest sample size. Table 5-5 characterises all data points (individual-specific mean WTP) according to the significant local cluster types to which individuals belong and identifies those individuals who are not part of any locally significant cluster. From this table, we can see that we found no cases of significant outliers in which a high value is surrounded primarily by low values (HL), and vice-versa (LH).

From table 5-5 we can also infer that the total number of HH identified in the pooled dataset (74) is smaller than the total number of LL (80). While analysing the site-specific datasets, we can only extrapolate the trend of having a higher number of HH (over LL) when the improvements occur in the Forth catchment area. This finding is interesting and emphasises the relevance of developing multi-scale studies when exploring spatial patterns in WTP (Johnston et al., 2015), as different patterns might only emerge when the scale of analysis is amplified (see annex 19).

Table 5-4 Global spatial autocorrelation of WTP estimates for ES improvements

	All		Tay		Clyde		Forth	
Attribute	Moran's I	P values	Moran's I	P values	Moran's I	P values	Moran's I	P values
Flood control								
Slight improvement	0.03	(0.00) **	-0.01	(0.66)	-0.02	(0.78)	-0.01	(0.69)
Large improvement	0.00	(0.54)	-0.04	(0.97)	-0.01	(0.51)	0.01	(0.16)
Biodiversity								
Slight improvement	0.00	(0.40)	0.00	(0.45)	-0.01	(0.65)	0.00	(0.37)
Large improvement	-0.01	(0.81)	0.00	(0.45)	-0.01	(0.63)	0.00	(0.37)
Recreation								
Slight improvement	-0.01	(0.86)	-0.01	(0.57)	0.01	(0.26)	-0.01	(0.52)
Large improvement	-0.01	(0.80)	-0.02	(0.75)	-0.04	(0.95)	0.01	(0.18)

The values in parenthesis are p-values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.

Table 5-5 Number of local clusters of WTP estimates for ES improvements

	HH				LL				HL				LH				NS			
Attribute	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay
Flood control																				
Slight improvement	24	5	2	4	26	1	5	3	0	0	0	0	0	0	0	0	521	182	187	182
Large improvement	13	9	3	1	17	3	11	0	0	0	0	0	0	0	0	0	541	176	180	188
Biodiversity																				
Slight improvement	7	2	4	2	10	1	11	0	0	0	0	0	0	0	0	0	554	185	179	187
Large improvement	10	4	11	3	7	1	6	10	0	0	0	0	0	0	0	0	554	183	177	176
Recreation																				
Slight improvement	7	6	2	6	6	1	1	2	0	0	0	0	0	0	0	0	558	181	191	181
Large improvement	13	0	7	6	14	2	8	2	0	0	0	0	0	0	0	0	544	186	179	181

HH=high-high, LL=low-low, HL=low-high, LH=high-low, NS=not significant; using k= 23 for neighbourhood definition.
The sum of the number of clusters associated with the same dataset (e.g. ‘All’) per row leads to the total sample (n=571).

Since we are dealing with different sample sizes for each survey, we used figure 5-2 together with the table in annex 19 which scales the numbers in proportion to sample size to refine the analysis. Even if the percentage of membership to any significant local cluster is not greater than 6% of the full sample (for all attributes, levels and datasets) our results reveal the presence of hotspots and coldspots of WTP for estuarine ES improvements in Scotland. The presence of significant local clusters suggests that respondent's preferences interact with their immediate spatial context and that they might feedback from the environmental, socioeconomic and cultural features of the local setting.

Figure 5-2 is a visual representation of the table presented in annex 19, which presents the same information as table 5-5 but in a percentage format. Both outputs were analysed together, and findings are as follows:

We found that there are differences in the numbers and the spatial distribution of significant hotspots and coldspots among all ES, across estuaries and for all levels of ES improvements. For the pooled sample (columns denoted by 'All' in annex 19) the percentage of a significant number of LL (14%) is slightly above the percentage of hotspots (13%). When comparing the patterns across surveys (see rows 1 to 3 in figure 5-2) we found that no survey sample repeatedly has the highest percentage of HH of WTP for improvements in all ES. In the Clyde sample (see row 1 in figure 5-2 or 'Clyde' columns in annex 19), for instance, we found the largest percentage of 'groups of neighbours' with high WTP for flood control improvements (7%). Whereas in the Forth sample (see row 2 in figure 5-2 or 'Forth' columns in annex 19) the largest percentage of HH was associated with biodiversity (8%). Finally, the Tay sample (see row 3 in figure 5-2 or 'Tay' columns in annex 19) had higher percentages of HH for improvements in recreation (6%).

The comparison of local clusters between surveys revealed that the largest percentage of groups of neighbours with low WTP (or LL) is associated with the Forth sample regardless of the estuarine ES in question (see row 4-6 in figure 5-2 or the second 'Forth' column in annex 21). On the other hand, while comparing the percentages of local clusters within surveys, we found that the highest percentages of the Clyde survey (see row 1 in figure 5-2) are related to flood control hotspots (with 7%). Regarding the Forth sample, we found that the two highest percentages of significant clusters are related to both the HH (8%) and LL (9%) of biodiversity improvements (see row 2 and 5 in figure 5-2 or annex 20). The latter finding reveals a somehow polarised trend, with important disparities on the ways in which the 'groups of neighbours' value the improvements on the provision of the same ES.

The membership likelihood of HH (or LL) might be partially explained by the relative distances to the area, as the HH of WTP are commonly contained in the catchment area analysed, while the coldspots of WTP are mostly located outside these areas (see figure 5-2 or annex 21). Testing this hypothesis formally is difficult to do with our sample size. However, we developed an exploratory analysis in annex 22 which uses one way ANOVA to evaluate whether significant differences in individual-specific WTP estimates exist at different distances to the area. We found that the differences were significant for slight improvements in flood control (F1).

The significant local clusters of WTP (HH and LL) were also plotted with a selected number of socioeconomic indicators to assess whether their spatial arrangement follows the spatial distribution of the population sociodemographic characteristics (see annex 23). Even though it is not possible to draw conclusions regarding the variables correlation strength from these maps, there seems to be an overlap of hotspots with the data zones having higher percentages of older people and females. This finding is in agreement with the ASC analysis which indicated that both, female and older people, have a significant preference for ES improvements.

Finally, when comparing the spatial distribution of positive and negative clusters in figure 5-2, we can identify a common trend for each cluster category. The hotspots of WTP estimates are mostly situated in densely populated areas in Scotland such as the Central Belt of Scotland (comprising the cities of Edinburgh and Glasgow) and the region close to Aberdeen. The spatial distribution of LL of WTP estimates is rather more complex. Coldspots for estuarine ES improvements are scattered in space, but they are frequently located in less populated regions such as the Highlands and the Islands. Our findings are contrasting to those of Campbell et al. (2009), who found that larger centres of populations led to lower WTP estimates for rural landscape improvements in Ireland.

However, this might be explained by the differences in the environmental management policies proposed. For our study, these consist of a restoration project impacting the environmental quality of Scottish cities directly as they are located inside the potentially restored catchment areas, whereas in Campbell et al. (2009) the proposed policy focus on providing environmental improvements in rural regions of Ireland, further away from urban centres.

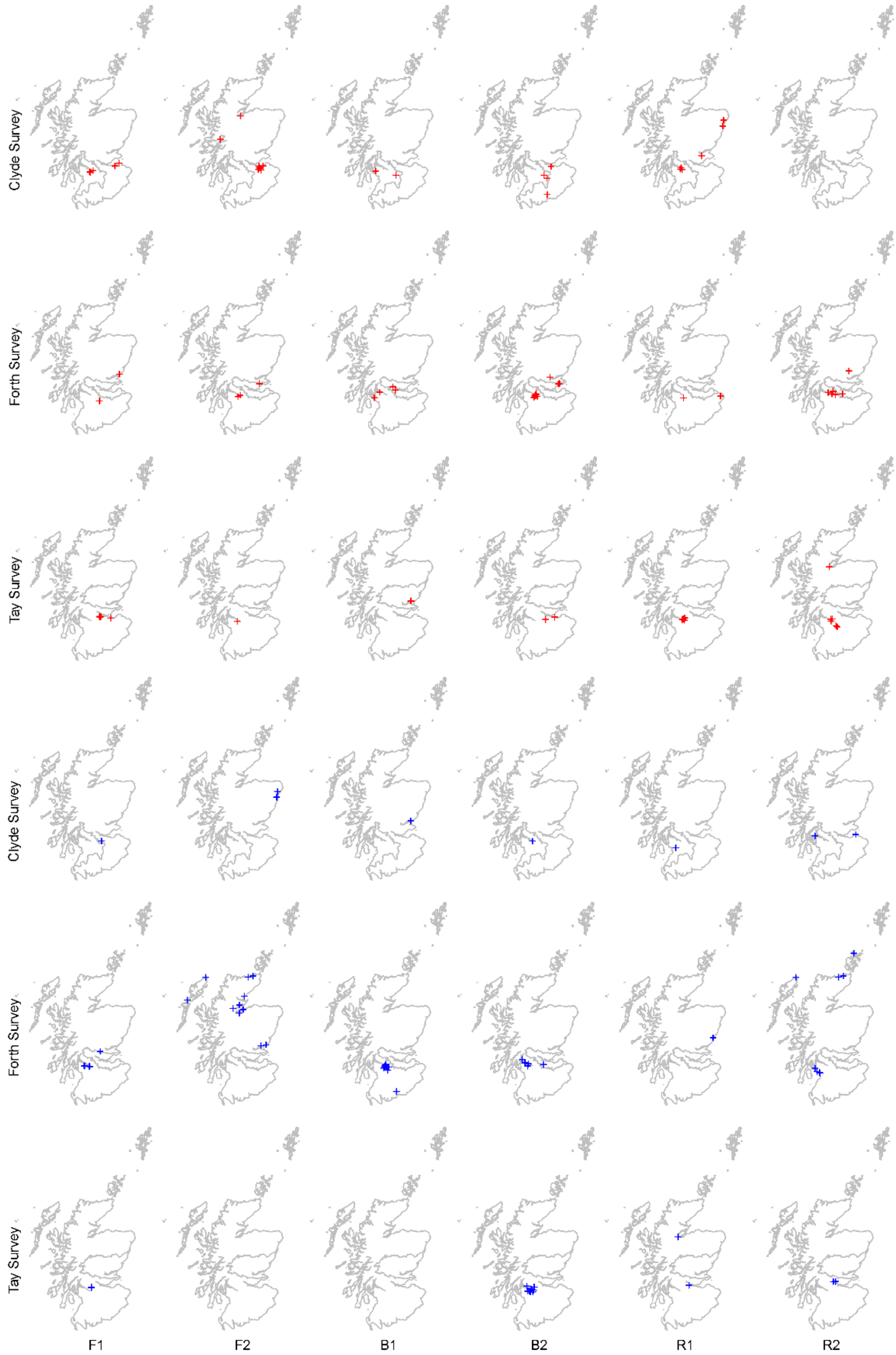

Figure 5-2 HH and LL of WTP for ecosystem services improvements

The red points represent HH and blue points LL. The letter case refers to the ES (flood=F, biodiversity=B, recreation=R), whereas the number refers to the level of improvement (1=slight improvement, 2=large improvement). Each map contains the catchment area polygon associated with its respective survey.

5.3.4. Comparative analysis of the geographical pattern of local clusters of willingness to pay estimates

The Ripley K and L functions have been used to generate policy recommendations related to agri-environmental (Bamière et al., 2013), farming (Bamière et al., 2008), forestry (Li and Zhang, 2007) and pest management policies (Lynch and Moorcroft, 2008). However, these functions have not yet been integrated with environmental valuation data to inform the management of estuarine ES.

This study developed a cross-type spatial point pattern analysis to explore whether the distribution patterns of significant hotspots (and coldspots) of WTP are similar among estuarine ES and across case studies. The K and L cross-type functions were estimated for HH and LL processes, independently. Results of the K-cross functions are in line with the findings derived from the L-cross function. Since the interpretation of the L cross-type function is more straightforward and the results are similar for all possible combinations of point patterns, we only present the figures plotting the $L_{Clyde,Forth}$ and $L_{biodiversity,flood}$ cross-type functions in the main text. Please refer to annex 24 and 25 to find the remaining L_{ij} cross-type plots, as well as annex 26 and 27 to review all the K_{ij} cross-type plots.

The plots in the first two rows of figure 5-3 and figure 5-4 layout the observed cross L function together with the theoretical Poisson L function, independently for LL and HH (see row 1 and 2, respectively). The remaining plots in these figures are used to test of the null hypothesis of CSRI between point types, for which they add the maximum and minimum L_{ij} over the 1000 simulation datasets to depict the upper and lower bound of the envelopes.

Figure 5-3 presents the pairwise comparison of one combination of the ‘survey-types’ point patterns and displays independent analysis for the points classified as HH (see row 1 and 3) and LL (see row 2 and 4). Figure 5-4 is organised similarly, but instead, it displays the K-cross plots for one combination of the ‘ES-types’. In both, figure 5-3 and figure 5-4 (as well as in annex 24 and 25), it can be seen that the L-cross function is outside the simulation envelope for almost every distance band (denoted by r), when referring to hotspots as well as coldspots.

This finding suggests the existence of ‘inter-ES’ clustering of HH (or LL) points for all distances, in addition to ‘inter-study’ clustering of HH (or LL) points for all distances. Put another way: i) the hotspots (or coldspots) of WTP for improvements in flood control, biodiversity and recreation commonly occur close to each other in space, ii) the hotspots (or coldspots) of WTP for improvements in estuarine ES delivered with a restoration project happening at the Clyde, Forth and Tay catchment area also tend to occur close to each other in space.

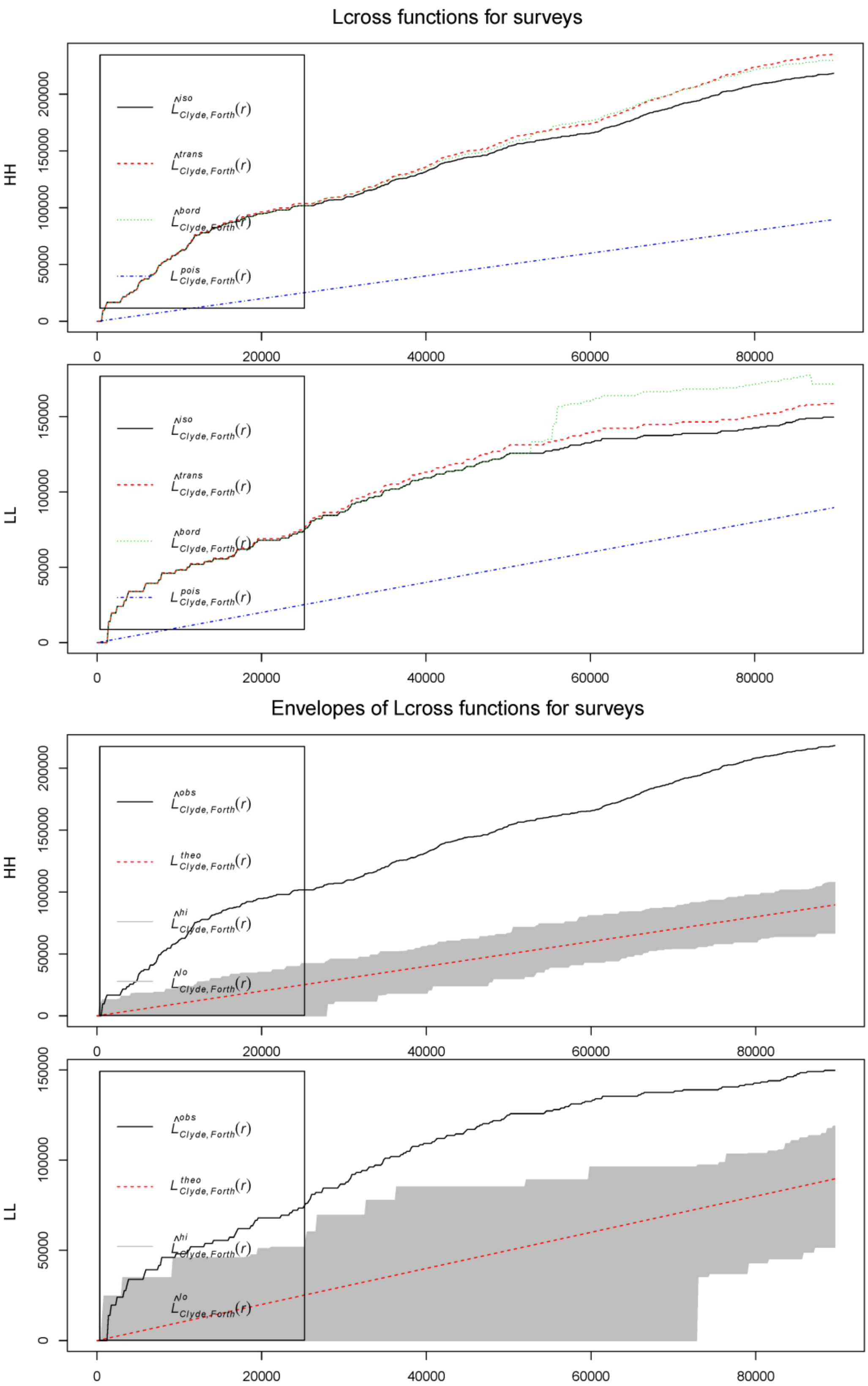

Figure 5-3 L-cross functions and envelopes for local clusters of WTP estimates marked by survey

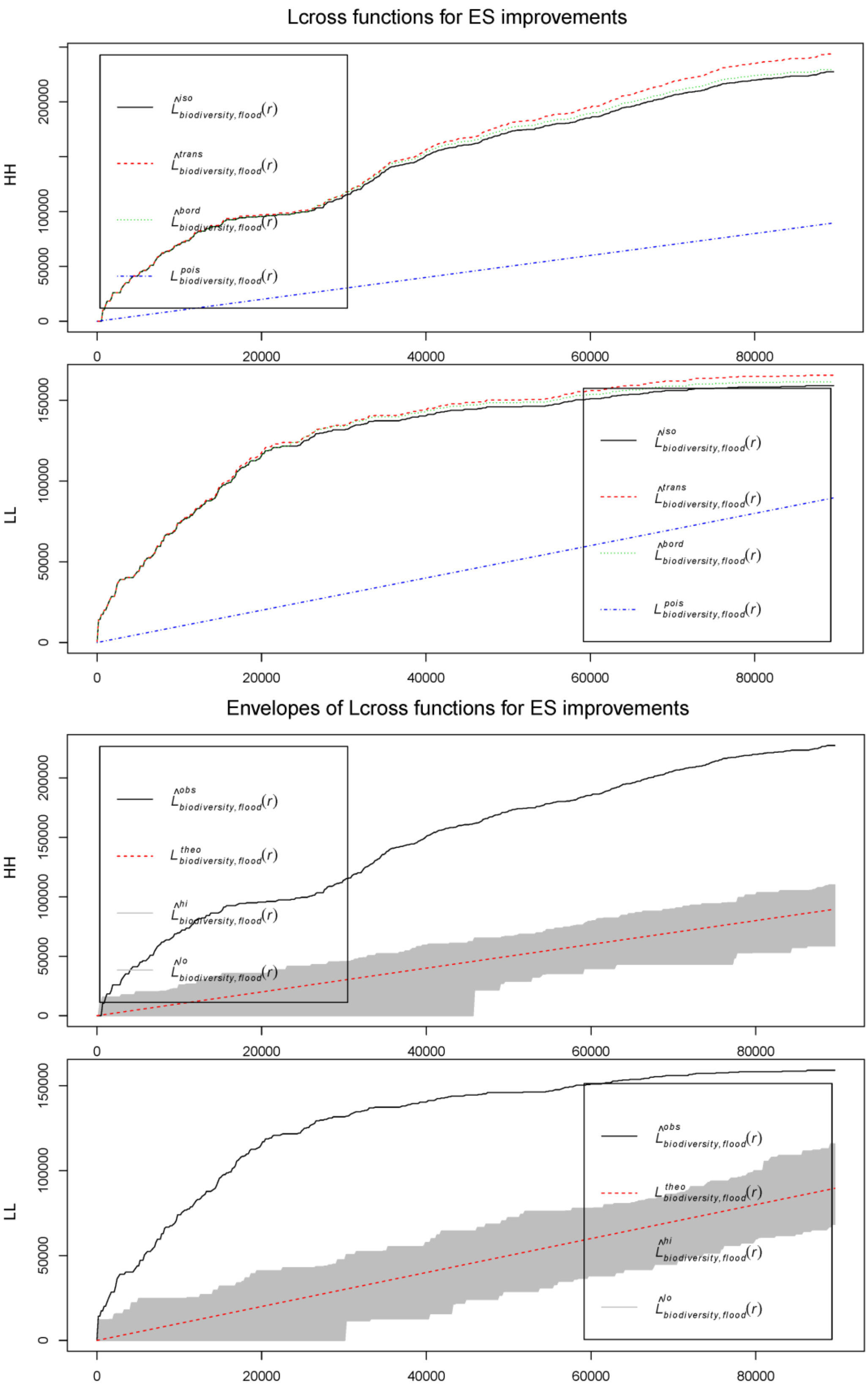

Figure 5-4 L-cross functions and envelopes for local clusters of WTP estimates marked by ES

5.4. Conclusions and policy implications

Overall our study outcomes support previous research recommendations (Johnston et al., 2011; Meyerhoff, 2013) which note the benefits of exploring local clustering of WTP estimates, even if distance decay or global patterns of spatial autocorrelation are not found. Our findings reveal that the geographical distribution of WTP for environmental goods are far more complex than those indicated by previous distance decay studies (Bateman et al., 2006; Concu, 2007; Hanley et al., 2003; Schaafsma et al., 2012), and support recent claims for using non-linear approaches of analysis which reveal ‘patchy’ patterns of clustering for environmental values (Johnston et al., 2011; Johnston and Ramachandran, 2014; Meyerhoff, 2013).

Conducting a spatially explicit analysis provides decision makers with some of the necessary information to develop locational targeting of policy interventions, and to enhance their efficiency in mitigating environmental degradation (Bateman et al., 2016). Similarly, the exploration of WTP variation at the individual level potentially helps to calibrate the design of environmental taxes (Yao et al., 2014). Thus, the consideration of irregular patterns of environmental values in environmental policy making demands the design of spatially explicit policies, which could benefit from using differential taxes in regions delimited in terms of their WTP for environmental improvements (or density of local clusters of WTP estimates).

Thinking spatially while generating environmental management plans is essential for creating efficient and optimal policies which take into consideration the spatial allocation of natural resources, together with the distribution of wealth (Czajkowski et al., 2015). Moreover, generating studies that reveal the spatial heterogeneity of environmental preferences at a fine scale helps to increase the spatial context awareness of environmental policy makers.

Downscaling the analysis of environmental preferences helps to increase the precision on which the values people place on environmental improvements to be understood, whether these improvements occur within their local neighbourhood, or outside. However, it also increases the degree of complexity of the information provided for guiding the design of environmental policies.

The present analysis provides evidence to confirm that hot (cold) spots of WTP for improved provision of estuarine ES are distributed similarly in space regardless of the ES in question, or the case study estuary in consideration. By contrasting the spatial distribution of local clusters of WTP estimates for various estuarine ES and case studies, our study demonstrates commonalities among their geographical distribution, which may flow from the spatial distribution of socioeconomic characteristics important to variations in WTP, such as income (Barbier et al., 2017; Jacobsen and Hanley, 2009). There is a further need to assess the determinants of hotspots (or coldspots) of WTP estimates to progress on the understanding of clustering patterns of environmental preferences. While this goes beyond the scope of the current study and might be difficult to achieve with the given survey sample, the author intends to develop this analysis in a further study.

These similarities among WTP could also be taken into consideration for summarising the information provided to policy makers, and to find the regions to target (prioritise) when developing restoration projects that aim to deliver improvements in estuarine ES provision levels. For instance, future studies could generate overlap analysis to identify regions with a higher density of HH of WTP for different environmental improvements, or model the probabilities of having clusters of high WTP across space for a set of potential environmental improvement projects.

The present analysis has explored how the heterogeneity of WTP estimates relates to the local geographic context. Environmental valuation studies often provide policy recommendations while ignoring the relevance of psychological factors such as attitudes, social norms, perceptions and beliefs on the process of environmental preference formation (Hanley et al., 2006; Hynes et al., 2013b). Therefore the authors develop this analysis in the following chapter.

Chapter 6. Attitude effects on preference heterogeneity

6.1. Introduction

A growing body of literature in environmental valuation makes use of ‘hybrid’ choice modelling frameworks to acknowledge the influence of psychological factors such as attitudes, social norms, perceptions and beliefs in individual decision making processes (Faccioli et al., 2018; Hess and Beharry-Borg, 2012; Mariel et al., 2015; Mariel and Meyerhoff, 2016). Estimating choice models in a hybrid framework has led to improvements in modelling individuals’ behaviour, and allayed concerns over endogeneity and *measurement bias* associated with the use of attitudinal variables (Czajkowski et al., 2017b; Daly et al., 2012). However, the capacity of the approach to explain underlying sources of heterogeneity linked to environmental perceptions and attitudes has not been extensively explored in the literature.

This chapter presents an empirical analysis addressing the *Specific objective 3* and answering the questions derived from it. As explained in chapter 1, the third empirical analysis aims to assess the role of environmental attitudes as an underlying source of preference heterogeneity regarding policies restoring estuarine ES. Chapter 2 has already explained why environmental attitudes, could be a potential source of preference heterogeneity. Thus this chapter investigates how environmental attitudes could impact individuals’ support for policies restoring estuarine ES.

Developing further understanding of the links between environmental attitudes and preferences is useful for the design of environmental policies. For instance, it allows policy makers to understand whether and to what extent promoting PEB could be achieved by means of changing environmental attitudes. Moreover, it provides further insight into the attitudinal motives and barriers to PEB. Therefore, studying the effect of environmental attitudes on preference heterogeneity for estuarine ES might be helpful in identifying what policies should be targeting when the aim is to promote pro-environmental attitudes and encourage sustainable behaviour in society.

This chapter, in conjunction with the analysis in chapters 4 and 5, aims to generate information to guide policy makers and regulators in designing more efficient and contextualised environmental management policies. The following analysis examines the influence of a latent environmental consciousness variable on respondents' attitudinal answers and their choices for environmental management alternatives portraying different improvements in estuarine ES. We used joint likelihood maximisation to estimate a HMXL model (Czajkowski et al., 2017b), which is a specific type of HCM including random parameters. To the best of our knowledge this is the first application of an HCM to the ES framework, and more importantly, this is the first attempt to explore whether preferences for the restoration of estuarine ES vary across individuals with a different degree of environmental consciousness.

The rest of this chapter is organised as follows. First, section 6.2 describes the hybrid choice modelling framework used to analyse the choice data. Subsequently, section 6.3 presents and discusses the results of the econometric models and the welfare analysis focused on WTP estimates. Finally, in section 6.4 we present a synthesis of the main findings and discuss their policy implications.

6.2. Empirical analysis

We used data from a DCE conducted in Scotland in 2016, which also collected six attitudinal statements describing respondents' awareness, knowledge, beliefs and concern regarding the degradation of estuarine ES. Details regarding the DCE design can be found in chapter 3. As it was explained in table 3-7, the analysis of this chapter uses the choice responses of a sample of 473 individuals. This sample was reached after deleting protest bid individuals, individuals with missing income statement or postcode information, as well as those respondents who declared having no formed opinion regarding the environmental attitudinal questions (i.e. who answered "I don't know" to these items). As in previous chapters, we applied t-test to determine that the sample is representative of the Scottish population in most of the available statistics, except age. The summary statistics for the sample used in this chapter are depicted in annex 28.

Several specification forms were tested for the HMXL model. Similarly, to the models in previous chapters, the better-fitted model defined the utility as a linear function of dummy coded attributes of that scenario and the ASC. The hybrid model was only estimated with the largest sample, which has been referred to in previous chapters as the pooled dataset.

Table 6-1 describes the coding used in the HMXL model, which was coded and estimated in R software (version 3.3.2).

Table 6-1 Explanation of variable abbreviations and coding

Variable	Explanation
ASC	Constant term (0 = Option1: NO new policy, 1 = Option 2 or 3)
F1	Change in flood control from "increase in flood risk" to "slight reduction in flood risk" (1 = yes, 0 = no)
F2	Change in flood control from "increase in flood risk" to "large reduction in flood risk" (1 = yes, 0 = no)
B1	Change in biodiversity from "decrease in biodiversity" to "slight increase in biodiversity" (1 = yes, 0 = no)
B2	Change in biodiversity from "decrease in biodiversity" to "large increase in biodiversity" (1 = yes, 0 = no)
R1	Change in recreation from "decrease in recreation" to "slight increase in recreation" (1 = yes, 0 = no)
R2	Change in recreation from "decrease in recreation" to "large increase in recreation" (1 = yes, 0 = no)
Cost	Additional council tax payment
Resident	Whether respondent resides in the catchment area (1 = yes, 0 = no)
Visitor	Whether respondent visited the area for outdoor recreational activities in the last 12 months (1 = yes, 0 = no)
Female	Respondent's gender (1 = Female, 0 = Male)
Age	Respondent's age is above the average (1 = yes, 0 = no)
Graduate	Whether respondent has undergraduate and/or postgraduate education (1 = yes, 0 = no)
Income	Respondent's income is above the average for the sample (1 = yes, 0 = no)
Forth	Whether respondent answered the Forth questionnaire (1 = yes, 0 = no)
Tay	Whether respondent answered the Clyde questionnaire (1 = yes, 0 = no)
Clyde	Omitted variable related to questionnaire answered

6.2.1. Choice modelling

The hybrid choice modelling framework was used to identify underlying sources of preferences heterogeneity for estuarine ES and to understand how environmental consciousness influences respondents' choices for ES management. We tested a hybrid multinomial logit (HMNL) specification but then used a hybrid model including random parameters HMXL because it fitted our data better.

The following section (6.2.1) outlines the HMXL model structure and its use for integrating attitudes into choice models. As figure 6-1 shows, the 'hybrid choice model' is composed of two structural equations, one for the choice model and one for the latent variable model, as well as a group of measurement relationships. The measurement equations are used so that the latent variable becomes 'measurable' and the second structural equation serves to link the latent variable with the standard discrete choice model components. The dotted lines in figure 6-1 refer to the contribution of the error term; the dashed lines indicate measurement relationships and the complete lines show casual links between two constructs.

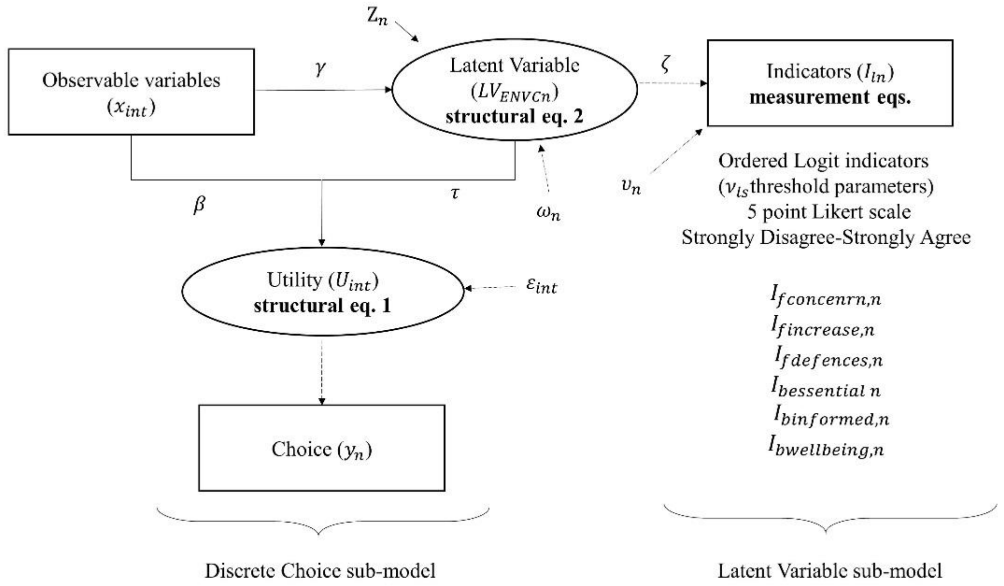

Figure 6-1 Outline of hybrid choice model structure

The first structural equation in the HMXL is based on the RUM theory which stipulates the indirect utility U_{int} that respondent n derives from alternative i in the choice occasion t is the sum of a deterministic and a random component. An individual's utility is given by:

$$U_{int} = V_{int} + \varepsilon_{int} \qquad 6\text{-}1$$

where ε_{int} captures the factors that affect utility but are not observed by the modeller and therefore not included in V_{int}. In our model, the deterministic component of utility is given by:

$$V_{int} = f\left(\beta_n, \tau, LV_{ENVC_n}, x_{int}, z_n\right) \qquad \text{6-2}$$

where β_n is a vector of tastes of respondent n, x_{int} is a vector of attributes of alternative i, τ is a vector of parameters that explain the impact of the latent variable LV_{ENVC_n} (specific to respondent n) on the utility of an alternative i (in interaction with the ASC coefficient), z_n is a vector of measured sociodemographic attributes of respondent n (possibly in interaction with the attributes x_{int}) and ε_{int} is an error term following an extreme value distribution with a location parameter 0 and scale parameter 1.

The variable LV_{ENVC} explain the answers of respondent n to a set of attitudinal questions relating to the concept of environmental consciousness. A hybrid structure is recommended to analyse the impact of these environmental attitudes on respondents' choices as the direct inclusion of the vector of these responses on the utility function V_{int} could lead to theoretically and statistically misguided results (Ashok et al., 2002; Ben-Akiva et al., 2002; M Ben-Akiva et al., 1999; Bolduc et al., 2005; Hess and Beharry-Borg, 2012). Attitudinal indicators are a function of underlying attitudes, rather than a direct measure of attitudes and they are likely to suffer measurement error. Their use as explanatory variables while ignoring their measurement error will lead to inconsistent estimation (Ashok et al., 2002). Moreover, responses to attitudinal questions could be correlated with unobserved factors entering the error term of the utility model and thus generate problems with *endogeneity bias* (Ben-Akiva et al., 2002; Moshe Ben-Akiva et al., 1999).

In order to recognise the latent nature of attitudes, the HCM uses the values of attitudinal indicators, as a dependent variable of a latent variable of environmental consciousness LV_{ENVC} rather than as direct explanatory variables of choice probability. This approach assumes that the underlying attitudes and perceptions of respondent n are described by the unobserved variable LV_{ENVC}. The latent variable therefore influences the answers that a respondent gives to questions of an attitudinal or perception nature (I_n) while also driving the behaviour in the actual choice situation.

$$V_{int} = ASC_{int} + \sigma_{ASC_{int}} \cdot \xi_{1,n} \cdot ASC_{int} + \tau_{LV_{ENVC}} LV_{ENVC_{int}} + \beta_{F1} \cdot \mathrm{F1} + \sigma_{F1} \cdot \xi_{1,n} \cdot \mathrm{F1}_{int} + \beta_{F2} \cdot \mathrm{F2} + \sigma_{F2} \cdot \xi_{1,n} \cdot \mathrm{F2}_{int} + \beta_{B1} \cdot \mathrm{B1} + \sigma_{B1} \cdot \xi_{1,n} \cdot \mathrm{B1}_{int} + \beta_{B2} \cdot \mathrm{B2} + \sigma_{B2} \cdot \xi_{1,n} \cdot \mathrm{B2}_{int} + \beta_{R1} \cdot \mathrm{R1} + \sigma_{R1} \cdot \xi_{1,n} \cdot \mathrm{R1}_{int} + \beta_{R2} \cdot \mathrm{R2} + \sigma_{R2} \cdot \xi_{1,n} \cdot \mathrm{R2}_{int} \quad \text{6-3}$$

where ASC is an alternative specific constant for the no policy alternative, taking the value of zero when respondents chose the *status quo* and one when they did not. After testing for several specification forms, we included all nonmonetary attributes into the utility function using a dummy coded specification.

Similarly to the RPL model used in chapters 4 and 5, the HMXL model includes random attribute coefficients that could follow a normal, lognormal, triangular, or uniform distribution. The model estimated in this chapter used a fixed (i.e. non-random) cost attribute and assumed normally distributed coefficients for the ES attributes and the ASC. The cost was assumed fixed to avoid convergence issues and to facilitate the implicit prices calculation (Revelt and Train, 1998; Wielgus et al., 2009). With this specification, $\xi_{1,n}$ is a random variate that follows a standard normal distribution across individual respondents, but is held constant across choices for the same respondent n. This ensures that the preference for those attributes now follow a Normal distribution across respondents, with mean β and standard deviation σ.

Environmental consciousness is hypothesised to be a function of an individual's socioeconomic characteristics, but at the same time is an explanatory variable in the measurement equations. We used one latent variable capturing underlying environmental consciousness. The structural equation for this latent variable is, therefore, given by:

$$LV_{ENVC} = h(\mathrm{z}_n, \gamma) + \omega_{ENVC} \quad \text{6-4}$$

where $h(\mathrm{z}_n, \gamma)$ represents the deterministic part of LV_{ENVC} , the specification $h()$ is in our case linear with z_n being a vector of sociodemographic variables of respondent n, and γ being a vector of estimated parameters denoting the structural relationship between the latent and observed variables. Additionally, ω_{ENVC} is a random disturbance which assumed to be normally distributed with a zero mean and a standard deviation σ_ω.Therefore, in our case, we have that:

$$LV_{ENVC} = \gamma_{resident} z_{resident,n} + \gamma_{visitor} z_{visitor,n} + \gamma_{female} z_{female,n} + \gamma_{\text{age}} z_{\text{age},n} + \gamma_{\text{graduate}} z_{\text{graduate},n} + \gamma_{\text{income}} z_{\text{income},n} + \gamma_{\text{forth}} z_{\text{forth},n} + \gamma_{\text{tay}} z_{\text{tay},n} + \omega_{ENVC} \quad \text{6-5}$$

where $z_{1n}, z_{2n}, \dots, z_{mn}$ are the specific sociodemographic variables and $\omega_n \sim N(0,1)$.

The measurement equations use the values of the attitudinal indicators as dependent variables. The lth indicator (of total 6 indicators) for respondent n is defined as:

$$I_{ln} = m(LV_{ENVC}, \zeta) + \upsilon_n \quad \text{6-6}$$

where the indicator $I_{ln}$is a function of the latent variable LV_{ENVC} and a vector of parameters ζ. The specification of $\upsilon_n$determines the behaviour of the measurement model and is dependent on the nature of the indicator.

The responses to the attitudinal statements or indicators were collected using a 6-point Likert scale going from “Strongly Disagree” to “Strongly Agree” which also differentiates between the “Neutral” and “I don’t know” categories. Individuals with “I don’t know” responses were removed to afterwards recode the answers in the commonly used 5-point Likert scale.

We recognised the ordered nature of the attitudinal questions $I_1$to I_6 by making use of an ordered logit structure. The measurement equations are therefore given by thresholds functions. For a discrete indicator with S levels $i_1, i_2, \dots, i_S$ such that $i_1 < i_2 < \cdots < i_S$, the measurement equation for individual n is modelled as an ordered logit model for the latent variable, where $\nu_1, \nu_2, \dots, \nu_{S-1}$are thresholds that need to be estimated:

$$I_{ln} = \begin{cases} i_1 \; if -\infty < LV_{ENVC} \leq \nu_{l.1} \\ i_2 \; if \; \nu_{l.1} < LV_{ENVC} \leq \nu_{l.2} \\ \vdots \\ i_s \; if \; \nu_{l.(S-1)} < LV_{ENVC} \leq \infty \end{cases} \quad \text{6-7}$$

Let $L(I_{ln} | LV_{ENVC}, \zeta_I, \nu_I)$ give the probability of observing the specific responses given by respondent n to the various attitudinal questions. The likelihood of the specific observed value of I_{ln} $(l = 1,2, \dots 6)$ is given by:

$$P_{I_{ln}} = L_{I_{ln}} = \sum_{s=2}^{S-1} I_{(I_{ln}=\, i_s)} \left[\frac{e^{\nu_{l.s}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.s}-\zeta_l LV_{ENVC}}} - \frac{e^{\nu_{l.(s-1)}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.(s-1)}-\zeta_l LV_{ENVC}}}\right] \tag{6-8}$$

where ζ_l measures the impact of the latent variable LV_{ENVC} on indicator I_l, and where $\nu_{l.s}$, $s = 0, \dots, 4$ are a set of estimated threshold parameters. For normalization, we set $\nu_{l.4}$ to $+\infty$, and $\nu_{l.0}$ to $-\infty$.

We then have

$$L_{I_{ln}} = I_{(I_{ln}=\, i_1)} \left[\frac{e^{\nu_{l.i_1}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.i_1}-\zeta_l LV_{ENVC}}}\right] + \sum_{s=2}^{S-1} I_{(I_{ln}=\, i_s)} \left[\frac{e^{\nu_{l.s}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.s}-\zeta_l LV_{ENVC}}} - \frac{e^{\nu_{l.(s-1)}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.(s-1)}-\zeta_l LV_{ENVC}}}\right] + I_{(I_{ln}=\, i_S)} \left[1 - \frac{e^{\nu_{l.(S-1)}-\zeta_l LV_{ENVC}}}{1+e^{\nu_{l.(S-1)}-\zeta_l LV_{ENVC}}}\right] \tag{6-9}$$

where ζ_l measures the impact of the latent variable LV_{ENVC} on indicator I_{ln} and $\nu_{l,1}, \nu_{l,2}, \dots, \nu_{l,(S-1)}$are estimated using a set of auxiliary parameters $\delta_{l,1}, \delta_{l,2}, \dots, \delta_{l,(S-2)}$such that:

$$\begin{aligned} \nu_{l,2} &= \nu_{l,1} + \delta_{l,1} \\ \nu_{l,3} &= \nu_{l,2} + \delta_{l,2} \\ \nu_{l,4} &= \nu_{l,3} + \delta_{l,3} \\ &\vdots \end{aligned} \tag{6-10}$$

where $\delta_{l,s} \geq 0 \ \forall \ s$. The definition of the auxiliary parameters assures that $\nu_{l,1} < \nu_{l,2} < \dots < \nu_{l,(S-1)}$.

Independent of the approach used for individual indicators (including the use of a mix of approaches), we can now write the probability of the observed set of respondent provided answers as:

$$P_{I_n} = \Pr\left(I_n^t \middle| \zeta_{I,} \Omega_I, LV_{ENVC}\right) = \prod_{l=1}^{L} P_{I_{ln}} \tag{6-11}$$

where $\zeta_{I,}$is a vector of estimated parameters showing the impact of the latent variable on the various indicators, and where Ω_I is a set of parameters relating to the specification of the measurement model, for example, standard deviations σ_I in the case of normal densities, or thresholds ν_I in the case of an ordered logit or probit specification.

The final component in the hybrid model is the choice model component. Let $L(y_n|\beta, \tau, LV_{ENVC}, z_n)$ give the likelihood of the observed sequence of T_n choices of respondent n (P_n) is the product of discrete choice probabilities depending on the model assumptions. In particular, we would have:

$$P_n = \Pr(y_n^t | \cdot) = \prod_{t=1}^{Tn} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{int}}} \tag{6-12}$$

In the simplest hybrid model, we would estimate a separate vector β_n for each respondent and set $\beta_n = \beta \; \forall \; n$, such that:

$$P_{int} = \Pr(y_n^t | \cdot) = \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{jnt}}} \tag{6-13}$$

However, as we allowed for additional random variations across respondents the choice probability for person n is given by:

$$P_{int} = \Pr(y_n^t | \cdot) = \int_{\beta} \frac{e^{V_{int}}}{\sum_{j=1}^{J} e^{V_{jnt}}} \, f(\beta|\Omega) d\beta, \tag{6-14}$$

where $\beta \sim f(\beta|\Omega)$ with the vector of parameters β and a covariance matrix Ω and $\beta \sim N(\mu|\sigma)$ with $\mu = 0$ and $\sigma = 1$ for the normal distribution.

The model is estimated by maximum likelihood. This estimation involves maximising the joint likelihood of the observed sequence of choices and the observed answers to the attitudinal questions simultaneously. The likelihood of the observed sequences has a specific form depending on the model assumptions, being $L\big(I_{ln}| \; \zeta_{I,} \; \nu_I, LV_{ENVC}\big)$ for our ordered logit model. On the other hand, the likelihood of the observed sequence of choices of respondent n (P_n) is the product of discrete choice probabilities depending on the model assumptions, which is $L(y_n|\beta, \tau, LV_{ENVC}, z_n)$ for the model used in the present work.

The two components are conditional on the given realisation of the latent variable LV_{ENVC}. Accordingly, the log-likelihood function of the HMNL model is given by the integration over ω_n:

$$LL\big(\beta, \gamma, \tau, z_n, \zeta_{I,} \nu_I\big) = \sum_{n=1}^{N} \ln \int_{\omega} L(Y_n| \cdot) L(I_n| \cdot) g(\omega_n) d\omega_n \tag{6-15}$$

Additional layers are added for the log-likelihood function of the HMXL model with independent random taste heterogeneity estimated in the present chapter:

$$LL(\Omega_\beta, \gamma, \tau, z_n, \zeta_{I,} \text{v}_I) = \sum_{n=1}^{N} \ln \int_\beta \int_\omega L(Y_n|\cdot) L(I_n|\cdot) g(\omega_n) f(\beta|\Omega) d\beta d\omega_n \tag{6-16}$$

That is expressed as,

$$LL(\Omega_\beta, \gamma, \tau, z_n, \zeta_{I,} \text{v}_I) = \sum_{n=1}^{N} \ln \int_\beta \int_\omega (P_n \prod_{l=1}^{L} P_{I_{ln}})\, g(\omega_n) f(\beta|\Omega) d\beta d\omega_n \tag{6-17}$$

Alternatively,

$$LL(\Omega_\beta, \gamma, \tau, z_n, \zeta_{I,} \text{v}_I) = \sum_{n=1}^{N} \ln \int_\beta \int_\omega P_n P_{I_n} g(\omega_n) f(\beta|\Omega) d\beta d\omega_n \tag{6-18}$$

where P_n is defined in equation 6-12, P_{I_n} is defined in equation 6-11, $P_{I_{ln}}$ is defined in equation 6-8 for $l = 1,2, \dots, 6$. The integration of the product of P_n and P_{I_n} over the distribution of β and ω is now required. This explains the presence of a density function for the random component in LV, i.e. ω and the density function for β, i.e. $f(\beta|\Omega_\beta)$. The latter is a function of an estimated vector of parameters Ω_β, while the parameters of the former have been normalised for identification (means to 0, variances to 1) as in Bolduc et al. (2005). The joint likelihood function thus depends on parameters of β distribution (Ω_β) and τ which capture the impact of the latent variable in the utility functions defined in equation 6-4, the vector $\gamma = (\gamma_0, \gamma_1, \gamma_2, \dots, \gamma_{m)}$ containing the parameters for the sociodemographic interactions in the latent variable specification defined in equation 6-5, $\zeta = (\zeta_1, \zeta_2, \dots, \zeta_{6)}$ and $\nu = (\nu_{1,1}, \nu_{1,2}, \dots, \nu_{1,(S-1)}, \dots, \nu_{6,1}, \nu_{6,2}, \dots, \nu_{6,(S-1)})$ defined in equation 6-6 and equation 6-7, respectively.

In practice, the joint likelihood equation does not possess a closed form solution. Simulation-based estimation of the model is used to evaluate P_n and P_{I_n} at a large number of draws from β and LV. The HCM literature recognises that the complexity of their design does not allow for using a high numbers of draws (Ben-akiva et al., 2002; Mariel and Meyerhoff, 2016), as this would result in a considerable increase in the estimation costs. As in practice, we simulated the random parameters distribution with 500 Halton draws. Halton-type draws were chosen over the Sobol draws used in previous chapters, as they are considered to be more accurate when using fewer draws (Bhat, 2003; Daziano and Bolduc, 2011). Additionally, we test if the model converges to the same solution (global maxima) by employing different starting values for the parameters in several estimation runs. The model was coded and estimated simultaneously using R software version 3.3.2. Simultaneous estimation is suggested by Bolduc & Alvarez-daziano (2010) to obtain efficient and consistent parameter estimates.

6.3. Empirical results and discussion

Our survey instrument included questions measuring individuals' environmental attitudes and beliefs using a 5-point Likert scale (five equates stronger agreement). The selected statements are used as indicators to explain the latent attitude of being *environmentally conscious*. Table 6-2 shows the indicators used and presents some summary statistics describing respondents' environmental attitudes. We used 'specific' attitudinal statements as it has been shown that they have a greater effect on preferences than statements describing more 'general' attitudes toward the natural environment (Faccioli et al., 2018).

Table 6-2 Responses to environmental attitudinal questions

Statement	Agreement in %
Attitude towards flood risk	
I am concerned about flooding *(fconcern)*	57.72
The frequency and extent of flooding are increasing where I live *(fincrease)*	27.48
I am worried that the current flood defences are not adequate enough to protect my home *(fdefences)*	24.31
Attitude towards biodiversity	
Biodiversity is essential for the production of goods such as food or fuel *(bessential)*	73.78
I am informed about biodiversity issues *(binformed)*	30.66
My well-being and quality of life depend on the area's biodiversity *(bwellbeing)*	39.75

Six-digit response scale: Strongly disagree, Disagree, Neutral, Agree, Strongly agree, I don't know. Agreement means agree or strongly agree.

Table 6-2 summarises the responses to the attitudinal questions. Half of the statements explore individuals' attitudes and beliefs towards flood risk, whereas the rest relate to biodiversity. Although low percentages of people (27%) thought that the frequency and extent of flooding increase over time, there is a general concern about flood risk. This worry drives half of the respondents (57%) to declare they are concerned about flood risks, whilst 24% of respondents to believe that the current flood defences would not protect their home. Regarding biodiversity, we found that 31% of respondents consider themselves informed about biodiversity issues. A significant amount of respondents

(74%) believed that biodiversity is essential for the provision of other ES, yet only one-third (37%) agreed that biodiversity impacts their well-being and quality of life.

This study defines the ‘environmental consciousness’ attitude as a multi-dimensional construct. The statements in table 6-2 were chosen so that they cover two relevant aspects describing *environmentally conscious* individuals. First, we considered the *cognitive dimension* (Jiménez Sánchez and Lafuente, 2010; Schlegelmilch et al., 1996; Walker, 2013) which measures individuals’ information and knowledge about environmental issues (see statements 2, 4 and 5). Studying the *cognitive dimension* is relevant since an individual’s knowledge level could trigger personal behavioural norms, as well as promote pro-environmental values and beliefs (Jiménez Sánchez and Lafuente, 2010). Second, we analysed an *affective dimension* (Dunlap, 2002; Jiménez Sánchez and Lafuente, 2010; Schlegelmilch et al., 1996) which reflect concern for the environment (perceived environmental degradation) which is shaped by individuals’ personal beliefs and values (see statements 1, 3 and 6). The *affective dimension* of environmental consciousness was included in our analysis as it has been found to be related with a ‘moral obligation’ towards the environment and the willingness to assume costs derived from environmental policies (Jiménez Sánchez and Lafuente, 2010).

People agreeing with these statements (see percentages in table 6-2) are more likely to be *environmentally conscious*. These values, beliefs and knowledge might not only be reflected in their attitudes but also feed into PEB (Alwitt and Berger, 1993; Corraliza and Berenguer, 2000; Cottrrel, 2003; Ellen et al., 1991; Gadenne et al., 2011; Karp, 1996). That is why having a higher degree of environmental consciousness can lead to stronger preferences for moving away from the *status quo* or “a willingness to assume the personal cost derived from developing environmental policies” (Jiménez Sánchez and Lafuente, 2010, p. 736), such as a restoration project delivering improvements on the provision of estuarine ES.

6.3.1. Hybrid mixed logit

This chapter uses an HMXL model to explore the influence of environmental consciousness on respondent’s choices for estuarine ES management and to assess for its contribution to preference heterogeneity. The estimates for the different model components are presented in table 6-3.

The overall fit of non-hybrid models to the data cannot be directly compared to the HMXL, as the fit of latter relates to the choice data in addition to the explanation of the indicator variables. However, the log-likelihood related to the choice components of a hybrid model can be compared to its reduced form, which is a choice model with no latent variables and where the marginal choice probabilities are expressed as a function of the observable explanatory variables (see RPL in annex 29). We found that modelling respondents choices with a hybrid framework result in a slightly better-fitted model (log-likelihood increase by 0.49 units). The model fit improvements which happens when comparing this model to the reduced RPL model suggest that the preference heterogeneity is not only related to observable measures such as the socioeconomics but also is also explained by latent attitudinal variables, in this case, environmental consciousness.

Table 6-3 HMXL estimates for ES improvements

Number of individuals:	473.00
Number of observations:	2838.00
Log-likelihood (overall):	-6044.13
Log-likelihood (choice component):	-2230.15
AIC:	12196.25
BIC:	12517.60

Utility functions				Measurement equations			
	Coefficients		T-rat		Coefficients		T-rat
ASC_{SQ}	-1.56	***	-5.13	ζ_{fconcern}	1.36	***	8.47
β_{F1}	1.61	***	11.79	$\nu_{\text{fconcern,1}}$	-2.84	***	-9.11
β_{F2}	2.03	***	12.22	$\nu_{\text{fconcern,2}}$	-1.38	***	-5.03
β_{B1}	1.66	***	11.63	$\nu_{\text{fconcern,3}}$	0.16		0.58
β_{B2}	1.83	***	11.85	$\nu_{\text{fconcern,4}}$	2.45	***	7.60
β_{R1}	0.62	***	7.05	$\zeta_{\text{fincrease}}$	3.03	***	7.06
β_{R2}	0.63	***	6.53	$\nu_{\text{fincrease,1}}$	-2.61	***	-3.99
β_{Cost}	-0.02	***	-10.63	$\nu_{\text{fincrease,2}}$	0.68		1.22
σ_{ASC}	2.83	***	-8.91	$\nu_{\text{fincrease,3}}$	3.20	***	5.17
σ_{F1}	0.65	***	-4.02	$\nu_{\text{fincrease,4}}$	6.02	***	7.48
σ_{F2}	1.21	***	8.08	$\zeta_{\text{fdefences}}$	2.54	***	8.45
σ_{B1}	0.33		1.55	$\nu_{\text{fdefences,1}}$	-1.49	**	-2.95
σ_{B2}	0.86	***	-6.06	$\nu_{\text{fdefences,2}}$	0.75		1.57
σ_{R1}	0.21		1.45	$\nu_{\text{fdefences,3}}$	3.03	***	5.77
σ_{R2}	0.60	***	3.54	$\nu_{\text{fdefences,4}}$	5.52	***	8.49
$\tau_{LV_{ENVC}}$	-0.47	*	-2.34	$\zeta_{\text{bessential}}$	0.37	**	3.27
Latent variable specification				$\nu_{\text{bessential,1}}$	-3.41	***	-3.41
	Est		T-rat	$\nu_{\text{bessential,2}}$	-2.80	***	-12.77
$\gamma_{resident}$	0.21		1.62	$\nu_{\text{bessential,3}}$	-0.93	***	-7.13
$\gamma_{visitor}$	0.27	*	2.30	$\nu_{\text{bessential,4}}$	0.93	***	6.76
γ_{female}	0.10		0.85	$\zeta_{\text{binformed}}$	0.63	***	4.63
γ_{age}	-0.17		-1.50	$\nu_{\text{binformed,1}}$	-2.33	***	-11.03
γ_{graduate}	-0.08		-0.69	$\nu_{\text{binformed,2}}$	-0.44	**	-2.71
γ_{income}	0.00		1.07	$\nu_{\text{binformed,3}}$	1.11	***	6.18
γ_{forth}	0.15		1.21	$\nu_{\text{binformed,4}}$	2.92	***	11.97
γ_{tay}	0.13		1.01	$\zeta_{\text{bwellbeing}}$	0.87	***	6.09
				$\nu_{\text{bwellbeing,1}}$	-2.45	***	-10.49
				$\nu_{\text{bwellbeing,2}}$	-1.02	***	-5.09
				$\nu_{\text{bwellbeing,3}}$	0.79	***	3.76
				$\nu_{\text{bwellbeing,4}}$	2.79	***	10.87

Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels. Standard errors computed by the Delta method.

Regarding the utility function estimates we found that all the non-monetary marginal utility coefficients (β) are positive and highly significant, meaning that any increase in estuarine ES provision levels increases respondent utility. The highest value was placed on large improvements in flood control (β_{F2}), whereas the lowest value is attached to slight improvements in recreational services (β_{R1}). The standard deviations attribute coefficients (σ) reveal significant unobserved heterogeneity for the large improvements on the provision levels of all estuarine ES ($\sigma_{F2}, \sigma_{B2}, \sigma_{R2}$) and slight improvements in flood control (σ_{F1}). Both, the ES attribute coefficients (and standard deviation) exhibit a positive scope effect with smaller values for smaller gains in estuarine ES provision (F1, B1 and R1) and larger values for more substantial changes (F2, B2 and R2). The negative and significant cost coefficient reveals that respondents are price sensitive and therefore prefer low-cost management options when all other attributes remain constant. The significantly negative ASC suggests positive impacts on respondent utility if moving away from the *status quo*. Moreover, we found significant unobserved heterogeneity for this variable.

The ζ parameters in the measurement equations and for the six attitudinal questions are positive and significant, indicating that respondents with a higher latent environmental consciousness are more likely to agree with the statements relating to respondent's awareness, knowledge, beliefs and concern towards the degradation of estuarine ES. Thus, the present finding suggests that the indicators used are indeed describing the latent attitude 'environmental consciousness'.

Similarly to Daly et al. (2012) who applied a HCM to study travel behaviour, the threshold coefficients ν of this analysis present asymmetry and differences in scale between all statements. Even though the estimated thresholds cuts of the latent variable differ among attitudinal questions (i.e. $\nu_{\text{fconcern},1} \neq \nu_{\text{fincrease},1} \neq \nu_{\text{fdefences},1} \neq \nu_{\text{bessential},1} \neq \nu_{\text{binformed},1} \neq \nu_{\text{bwellbeing},1}$), overall, we found that moving from a lower to a higher threshold results in higher levels of the latent attitude (i.e. $\nu_{\text{fconcern},1} > \nu_{\text{fconcern},2} > \nu_{\text{fconcern},3} > \nu_{\text{fconcern},4}$). In other words, as people cross thresholds from disagreeing more strongly to agreeing more strongly with the attitudinal statements, their latent environmental consciousness increases.

In comparison to the standard RPL model, the HXML model provides additional policy-relevant information. For instance, it is possible to explore how sociodemographic factors relate to the latent attitude in the structural equation. A relevant finding from this equation is that the visitor variable is the only variable presenting a positive and significant correlation (at the 10% level) with the latent environmental consciousness attitude. Other variables, such as level of education or income, are not correlated with the latent variable. This finding suggests that people visiting the area for doing outdoor activities present a significantly higher latent attitude, i.e. they are more *environmentally conscious*.

The remaining γ estimates of the structural equations for the latent variable environmental consciousness are not significant, but suggest that the latent attitude is higher for residents, females, people with higher income and respondents answering the Forth and Tay questionnaire (in comparison to those answering the Clyde questionnaire). Lastly, we obtained unexpected signs for the remaining γ estimates suggesting that older and more educated respondents have a more negative latent attitude towards environmental improvements, although these estimates are insignificant at the 5% level. The failure to reach significance for the γ estimates confirms that, in general, socioeconomic variables are poor predictors of latent variables representing environmental attitudes and therefore we are analysing a 'truly latent' concept (Vij and Walker, 2016).

Finally, the interaction of the latent variable with the ASC reveals the existence of systematic heterogeneity regarding preferences for estuarine ES improvements. The τ estimate indicates a significantly negative impact (at the 10% level) of the latent variable on the ASC coefficient, meaning that more *environmentally conscious* individuals exhibit a stronger preference to change and to avoid the current state of degradation of ES. This finding is consistent with the research by Daziano and Bolduc (2011) who find that more *environmentally conscious* individuals are more likely to choose low-emission vehicles. Moreover, this results are in line with studies finding a link between environmental consciousness and ecologically conscious consumer behaviour (Roberts and Bacon, 1997).

6.3.2. Analysis of willingness to pay for ecosystem services

This section analyses the welfare measures computed with the HMXL estimates. Table 6-4 gives the amount of money that respondents are willing to pay for an increase in the ES provision levels, with respect to the baseline scenario of ES degradation. We applied the Krinsky and Robb (1986) bootstrap procedure with 1000 replications of the unconditional parameters to calculate the CI of the WTP estimates.

Results indicate that respondents have a positive WTP for policies that increase the provision of estuarine ES in Scotland. As table 6-4 shows, large improvements in flood control (F2) have the highest annual average WTP but also the widest CI. On the other hand, the WTP estimates for slight improvements in recreational services (R1) are the smallest and more precise (narrower CI). In agreement with the results of the previous empirical analysis (chapters 4 and 5), we found that the WTP estimates for flood control and biodiversity are at least 250% greater than both levels of recreational service enhancements. Finally, the welfare estimates confirm the presence of a positive scope effect in WTP estimates, which consist on smaller WTP estimates for minor gains in the provision of all ES, and greater WTP estimates for more substantial ES gains.

The WTP estimates obtained from the HMXL are similar to the ones obtained in previous empirical chapters (see table 4-7 and table 5-3). Furthermore, the estimates in table 6-4 are comparable to the welfare estimations obtained by Birol et al. (2009), who apply simpler CM frameworks to value ES improvements at the Bobrek wetland (in Poland) and report an annual WTP of £103 for biodiversity, £216 for flood control and £76 for riverbank access.[16]

[16] The values in GBP were calculated using the reported by OANDA (2018): 1 PLN ≈ 0.20 GBP.

Table 6-4 WTP estimates for ES improvements

	HMXL				
Attribute	WTP	C.I.			
Flood control					
Slight improvement	104.38	(	86.12	124.18	)
Large improvement	131.57	(	110.52	157.00	)
Biodiversity					
Slight improvement	108.05	(	88.98	128.60	)
Large improvement	118.90	(	99.56	141.00	)
Recreation					
Slight improvement	40.57	(	29.05	52.94	)
Large improvement	40.73	(	28.15	53.96	)

Unit GBP. Parenthesis indicate the size of the confidence interval.

6.4. Conclusions and policy implications

As Schubert and Chai (2012) have shown, there is a scope for generating policies which influence preference formation without violating consumer sovereignty. Policies that change environmental preferences could facilitate the transition to more sustainable paths of development (Brennan, 2006; Mattauch and Hepburn, 2016; Norton et al., 1998) by shaping society consumption patterns (Story et al., 2008; Weinberger and Goetzke, 2010) and influence the degree in which people behave altruistically (Bowles and Polania-Reyes, 2012).

In order to influence individual preferences towards estuarine management options that secure ES provision, there is prior need to understand all the factors that influence them. Generating further insights on the attitude-behaviour link can, for instance, help decision makers to understand whether the promotion of pro-environmental attitudes would enhance society's support for policies for restoring estuarine ES. In this way, investigating the psychological factors in individuals' decision making could be useful in developing more effective and socially acceptable environmental policies (Faccioli et al., 2018; Hoyos et al., 2015; Kim et al., 2014a).

The HCM framework is convenient to investigate the significance of psychological factors (e.g. attitudes, self-identity or personality) in decision making. Identifying the role of latent attitudes as motives or barriers for PEB adds a new layer of complexity, but at the same time, highlights the relevance of generating more positive attitudes as a prior step to promote proactive environmental behaviour. While the significance of attitudinal variables as determinants of environmental preferences has been studied before (see Milon and Scrogin, 2006; Solinõ and Farizo, 2014), the use of HCM accounts for the potential endogeneity of responses to attitudinal variables. Thus, using hybrid modelling frameworks thereby reduces the risk of developing biased estimates, as suggested by Daly et al. (2012).

The findings of the present analysis indicated that higher acceptance of policies restoring estuarine ES is associated with higher latent environmental consciousness. This finding supports environmental psychology theories suggesting that individuals with higher degrees of awareness, knowledge, and concern towards the degradation of ES tend to be more *environmentally conscious*, as well as more likely to choose natural resource management strategies that improve the ecologic quality of an area (Corraliza and Berenguer, 2000; Roberts and Bacon, 1997; Zelezny and Schultz, 2000). The analysis of this chapter provided empirical evidence to confirm that environmental attitudes, such as environmental consciousness, are a significant source of preference heterogeneity for improvements on estuarine ES provision in Scotland. Moreover, environmental consciousness was found to depend partly on whether respondents had visited the area for recreation, i.e. had direct interaction with the ecosystem in question.

The previous finding suggests that policy makers could promote PEB via the generation of attitudinal changes and that these, in turn, could be boosted through the promotion of outdoor recreational activities. The relationship between outdoor recreation participation and PEB has been identified in previous empirical studies (Barker and Dawson, 2012; Dunlap and Heffernan, 1975; Larson et al., 2011). Thus, we suggest that management plans aiming to restore estuarine ES could benefit from policies improving society's environmental consciousness, and they could do it through the development of local programmes of outdoor recreation education. We recommend restoration policies to be complemented by environmental and outdoor recreation education programmes which help to increase society’s environmental consciousness and might boost their engagement in pro-environment actions.

Previous studies have suggested that visitors commonly develop a sense of attachment and generate emotional bonds to a particular place when recreating in their natural areas (Hwang et al., 2005; Kaltenborn, 1997; Kyle et al., 2005, 2003). The multidimensional emotional bond, often called 'place attachment' is expressed in varying levels of pro-environment intentions and behaviour (Halpenny, 2010; Ramkissoon et al., 2013b), as well as an in an increase of their degree of environmental consciousness. Frequently engaging in outdoor recreational activities could improve respondent's degree of environmental consciousness by means of changes in its *cognitive* and *affective dimension*. For instance, respondents could become more knowledgeable and aware of environmental issues, as well as strengthen their emotional link with the natural environment as they develop life experiences in it. This, in turn, affects visitors preferences for environmental improvements and generates a higher willingness to fund policies that enhance estuarine ES provision levels (see results in chapter 4).

The present research could also be linked with the choice modelling literature that used behavioural theories and tested their effect on individuals' environmental preferences. For instance, some authors have used the Theory of Planned Behaviour (TPB) to predict individuals' environmental preference for conservation policies (Bernath and Roschewitz, 2008; Börger and Hattam, 2017). Similarly to our study, these researchers have found that attitudes, as well as subjective norms and the perceived behavioural control, predict individuals' choices for policies delivering environmental improvements.

The HCM developed in this chapter studied the *affective* (values and perceptions) and the *cognitive dimension* conditions (level of information) of environmental consciousness as defined by Jiménez Sánchez and Lafuente (2010). Future work might extend this analysis to also study the effect of the dispositional (personal attitudes) and the active dimension (pro-environmental behaviour) of this latent variable on choices for natural resources management plans. In addition to this, future use of standardised scales measuring individuals' degree of environmental consciousness as the New Ecological Paradigm (NEP) proposed by Dunlap et al. (2000) or Weigel and Weigel (1978) could facilitate the comparison of results across study cases, natural goods and ES.

Even though the hybrid choice modelling framework increases computational effort, and the latent variables they use might be associated with low levels of significance (see Boyce et al., 2017; Faccioli et al., 2018; Hoyos et al., 2015), we consider that the real gain that comes with modelling choices with this structure is the capacity to account for the multi-dimensionality of the decision making process. The generation of novel behavioural insights requires the use of more behaviourally realistic and holistic frameworks of analysis. The HCM thus is a useful platform to study and empirically test the psychology behind decision making and economic behaviour.

Part IV. Discussion

Chapter 7. General conclusion and discussion

Governments worldwide have expressed an increasing interest in the use of economic policy instruments, as opposed to the traditional forms of regulation for environmental protection (Aidt and Dutta, 2004; Atkinson et al., 2018). Environmental valuation studies have been informative in the process of designing environmental policies around the globe (Ascher and Steelman, 2006; Atkinson et al., 2018; Guo and Kildow, 2014; Laurans et al., 2013; Pearce and Seccombe-Hett, 2000). In the UK context, the applications of these studies have frequently been developed alongside the policy process (Atkinson et al., 2018; Hockley, 2014). For instance, the economic valuation of the environment has been integrated into the guidelines for developing environmental appraisals (Department of the Environment, 1991; Great Britain: Treasury, 2003; Pearce, 1998); have influenced the design of the Forestry Commission afforestation policies; and have been used in designing the landfill (CSERGE et al., 1993) and pesticides taxes (ECOTEC and EFTEC, 1999).

Regarding coastal and estuarine ecosystems, environmental valuation has played an important role in informing policy decisions for managing water quality in the UK (Atkinson et al., 2018). Furthermore, SP studies have been used to estimating the economic benefits resulting from implementing the EU Bathing Waters Directive (2006/7EC) (Hynes et al., 2013b) and the Water Framework Directive (EC 2000/60/EC) (Hanley et al., 2006; Metcalfe et al., 2012).

Against this background, this thesis estimates the welfare benefits of implementing a more integrative river basin management policy as proposed by Natural Scotland (2013b), which would result in an improvement of estuarine ES provision levels. We developed an application of a DCE in Scotland and utilised diverse CM approaches to explore different sources of heterogeneity in preferences for policies restoring estuarine ES. This study aims to inform decision makers about the barriers and motivations in developing catchment-based restoration projects to improve the levels of estuarine ES provision in Scotland. This final chapter comments on the potential use of this research, by explaining ways in which the obtained results could feed directly into environmental policies around estuarine and coastal management in Scotland.

This final chapter is organised as follows. The first section (7.1) presents a summary of the main research findings. Following this, we included a section (7.2) referring to the research achievements and limitations, which afterwards leads to the discussion of the directions that future research could take in section 7.3. The fourth section (7.4) touches upon the general policy implications of the research results, to finally present general concluding remarks in section 7.5.

7.1. Results summary

The present research developed a DCE to estimate individuals' WTP for restoring the provision levels of estuarine ES in Scotland. The empirical analysis utilised diverse CM approaches to explore different sources of heterogeneity in preferences for estuarine ES. The results obtained suggest that preference heterogeneity is partially explained by i) the ES type and ii) the catchment area in question; as well as respondent's iii) type of use and their iv) level of environmental consciousness. Finally, the v) local geographical context was also found to have a significant effect on preference heterogeneity.

Regarding the welfare estimates, this research found that Scottish citizens have a positive and significant WTP for improving the levels of flood control, biodiversity and recreation in the Clyde, Forth and Tay catchment areas. The hotspots of WTP for restoring any ES, in either catchment area, were found to be located in the more densely populated regions in Scotland. Additionally, we found that the WTP for recreational services were found to be lower (by at least a factor of two) on average than for either flood control or biodiversity conservation.

Even though estuarine recreational services were not ranked at the top amongst all ES, we found that the average WTP estimates were higher when respondents declared to have visited the study area for outdoor recreational purposes. Engaging in outdoor recreational activities was found to raise the level of environmental consciousness in individuals, which was in turn associated with higher levels of support for measures restoring estuarine ES. Finally, our results indicate that respondents from all over Scotland *assigned* higher WTP for restoring the catchment area which presently has the highest environmental quality (Tay). Conversely, the lowest WTP was associated with the restoration of the more environmentally degraded area contained within the Clyde catchment.

7.2. General discussion

Environmental valuation studies represent an opportunity for democratising environmental policies by allowing people to express their preferences towards environmental changes (Horne, 2006; Menegaki et al., 2007; Pearce and Seccombe-Hett, 2000; Stigka et al., 2014); and by assisting in the weighing-up of the benefits and costs to all of those affected by a policy change. The present research allowed Scottish citizens to express their preferences for policies restoring ES, and thus represent an opportunity for raising the acceptance of Scottish citizens of a plan for river basin management proposed by Natural Scotland (2013b).

Applications of DCE to value environmental goods and ES need to account for the effect of factors in addition to sociodemographic variables when analysing preference heterogeneity (Börger and Hattam, 2017; Boyce et al., 2017; Czajkowski et al., 2016; Drechsler et al., 2011; Faccioli et al., 2018; Kim et al., 2014a; Nielsen-Pincus et al., 2017). Our study contributes to the literature not only by augmenting the understanding of significant preference heterogeneity, but also by raising the awareness of the great variety of factors influencing individual decision making.

The analysis developed in this document consists of a novel attempt to consider a broader set of factors influencing environmental preferences to provide policy recommendations which consider their accumulated effect on welfare estimates. In this sense, our analysis attempts to generate policy guidelines while considering the simultaneous effect of sociodemographics, attitudes and the contextual space on welfare estimation. There are still very few environmental valuation studies using the modelling approaches applied in chapters 5 and 6. For instance, Ripley's functions have not been applied previously in the environmental valuation literature to compare spatial patterns of posterior WTP estimates (see sections 5.2.3 and 5.3.4). Moreover, there are very few applications of the HCM framework (see sections 6.2.1 and 6.3.1) which value ES (Faccioli et al., 2018; Hess and Beharry-Borg, 2012).

Our study highlights the need for using more flexible CM frameworks which permit the integration of additional layers of preference heterogeneity. The empirical results of chapters 4, 5 and 6, demonstrate that including further sources of heterogeneity results in log-likelihood and BIC improvements. However, as it can be seen in Table 7-1, this does not necessarily lead to more precise welfare estimates (narrower CI). Nonetheless, the slight reduction of the sample size between the models might explain why model estimates do not become more precise, so we recommend the use of larger samples when estimating complex CM.

Table 7-1 Model fit comparison

	Chapter 4		Chapter 5	Chapter 6
	MNL	RPL	Posterior RPL	HMXL
Log-likelihood (choice component)	-3140.18	-2755.44	-2662.65	-2230.15
F1 coefficients [1]	113.54	112.89	111.30	104.38
	(98.61-129.84)	(95.28-132.05)	(62.70-157.80)	(86.12-124.18)
F2 coefficients [1]	141.43	144.42	141.30	131.57
	(125.06-160.99)	(123.43-168.77)	(12.45-246.20)	(110.52-157.00)
B1 coefficients [1]	101.63	114.10	114.00	108.05
	(86.59-117.84)	(95.91-134.83)	(113.80-114.20)	(88.98-128.60)
B2 coefficients [1]	111.27	123.48	122.00	118.90
	(95.74-126.90)	(104.59-144.26)	(54.97-204.10)	(99.56-141.00)
R1 coefficients [1]	37.85	43.68	42.20	40.57
	(25.58-49.27)	(32.47-55.77)	(41.38-42.60)	(29.05-52.94)
R2 coefficients [1]	40.17	43.24	42.65	40.73
	(29.73-51.58)	(31.22-55.86)	(-10.35-100.90)	(28.15-53.96)
Observations	3534.00	3534.00	3426.00	2838.00
Adjusted rho-sq	0.19	0.29	0.29	NA
AIC	6296.36	5540.89	5367.29	12196.25 [2]
BIC	6345.72	5633.44	5496.21	12517.6 [2]

[1]Mean (lower and upper bound of CI).
[2]The overall fit of hybrid models cannot be compared with non-hybrid models as the statistic relates to the choice data in addition to the explanation of the indicator variables.

Overall, the results of the present thesis are in agreement with the previous literature and suggest that models which increase the realism of the decision making process yield gains as they explain individuals' decision making process better (Birol et al., 2006; Boyce et al., 2017; Campbell et al., 2009; Hess and Beharry-Borg, 2012; Mariel et al., 2015; Vij and Walker, 2016). The increase of modelling realism while estimating welfare estimates is not only beneficial from the modelling perspective, but also has real-world applications as it permits to develop better-informed designs of environmental policies (see sections 7.4 and 7.5).

Improvements in environmental valuation methods do not only focus on raising the accuracy of estimates, but also discuss the well-known problem of generalisability of study outputs (WTP estimates and policy recommendations) derived from environmental valuations research by developing benefit-transfer analysis (Bateman et al., 2006; Birr-Pedersen, 2006; Colombo et al., 2007; Hynes et al., 2013a; Ian et al., 1998; Nelson and Kennedy, 2009; Plummer, 2009; Wright, 2002). In the context of DCE studies, multi-case study analysis can be used to test the same hypothesis at different scales (i.e. regional vs national), as well as to develop a comparative approach to identify key factors influencing WTP estimates. Moreover, generating multi-case analysis can be used as a cross-validation method and a strategy to increase the generalizability of study outputs. The choice dataset used in the present study is unusual, as it permits us to explore preference heterogeneity across study sites, and among different ES. Therefore, our research contributes to the body of environmental valuation literature related to ES by developing an innovative approach for doing a comparative analysis of welfare estimates.

7.3. Further research

In order to advance the understanding of choice heuristics (Alemu et al., 2013; Amir and Levav, 2008; Campbell et al., 2011; Hensher et al., 2015; Leong and Hensher, 2012; Scarpa et al., 2009) it is recommended to take analysis of preference heterogeneity to a further level. Instead of developing independent analyses for each potential source of preference heterogeneity, as it was done in this study, future research should focus on developing new models which are capable of accounting for the effect of observable, latent and contextual variables simultaneously. There is an enormous potential for merging structural equations with choice models in the HCM framework proposed by (McFadden, 1986). For instance, few authors have augmented HCM by exploring the simultaneous effect of a mixture of latent influences (e.g. environmental attitudes, social influence, social environment) on environmental preferences (Kamargianni et al., 2014; Kim et al., 2014a). Further work could also generate latent variables describing the ‘environmental context’ at the individual level which could be measured through the use of GIS-based environmental quality indicators. Using the HCM in this way would allow to further understand to which extent local clustering of WTP is driven by similarities of preferences in a specific location, or instead, is driven by the influence of the ‘local natural context’ on society’s environmental preferences.

A growing body of environmental valuation literature argues for incorporating further relevant factors while estimating contingent values (Aldrich et al., 2007; Cunha-e-Sá et al., 2012; Meyerhoff, 2006; Sauer and Fischer, 2010; Spash et al., 2009) and modelling choice behaviour (Boyce et al., 2017; Campbell et al., 2009; Czajkowski et al., 2016; Faccioli et al., 2018; Hess, 2007; Kim et al., 2014a). In order to estimate more complex models studies need large and representative samples. The majority of the empirical applications of DCE to the environment attempt to obtain samples which are representative of the population in terms of socioeconomic characteristics. However few studies are concerned about achieving the spatial representativeness of the sample (Campbell, 2007; Campbell et al., 2009; Schaafsma, 2010; Schaafsma et al., 2012). It has been suggested that not using spatially representative samples could lead to bias estimates to quantify the aggregated welfare impact (Bateman et al., 2006). However, further analysis is needed to test whether obtaining spatially representative samples also help to improve the accuracy of individual-level WTP estimates, especially in studies aiming to include the spatial dimension into the analysis of environmental choices. Another consideration to explore in future studies is if the welfare estimates and the results of second-stage analysis could be biased/different when sampling only the residents of an area. This thesis attempted to explore this in chapter x, but it was inviable since the resident subset sample for each study site is very small.

Moreover, it has to be recognised that like any other survey-based research instrument, DCE has limitations on the amount of additional information to be collected without resulting in a heavy cognitive burden for respondents. We limited our analysis to the study of three potentially relevant sources of preference heterogeneity. However, it is the intention of the researcher to use the hybrid model framework to extend the preference heterogeneity analysis further to examine additional factors such as the temporal context (e.g. historical events), the social interaction (e.g. family and/or neighbours influence), individuals' levels of concern for the welfare of others (e.g. altruism), and perceived environmental quality (e.g. perceived abundance). Using more complex modelling structures could provide additional insights about the effect of altruism and social motivations (Bartczak, 2015; Cooper et al., 2004; Lee and Chung, 2012) or perceived environmental quality (Cameron et al., 2011; Domínguez-Torreiro and Soliño, 2011; Kataria et al., 2012; Leggett, 2002) on the decision making process, as well as increase the understanding about ways in which these factors are linked with respondents' socioeconomic characteristics.

More generally, two factors hamper the use of more 'complex' or 'advanced' CM techniques. First, their estimation commonly results in a substantial increase of the computational efforts as models could take several days to converge. Second, advanced choice models considerably increase coding efforts as they are not available in standard statistical packages. Czajkowski et al. (2017), have contributed to making these models available by developing a series of HCM and making the relevant Matlab code available with their publication.[17] Nonetheless, future efforts could be focused on making them more readily available to choice modellers in open source statistical packages.

Applied choice experiments also have temporal and financial limitations for delivering research outputs. The financial limitations of the present analysis, for instance, restricted the ways in which stakeholders could participate throughout the design process of the DCE. Although deliberative approaches to valuation (e.g. focus groups and visioning workshops) could help to empower citizens through the democratisation of the decision making process (Brown et al., 1995; Jacobs, 1997; Kenyon et al., 2001; Lo and Spash, 2013; Sagoff, 1998; Spash, 2001; Ward, 1999), it has also been suggested that the quality, significance and the legitimacy of valuation study outcomes are often dependent on ways in which participation is framed (Carnoye and Lopes, 2015; Jacobs, 1997; Niemeyer and Spash, 2001). Deliberative environmental valuation is considered to be more useful when individuals have a direct financial relationship with the agency proposed to collect the

[17] http://czaj.org/research/estimation-packages/dce.

funds (Niemeyer and Spash, 2001). For instance, when the environmental project involves local councils to which citizens already pay taxes, or organisations to which citizens already pay bills. Moreover, it has been found that individual participatory methods are more capable of generating transparent and quantifiable data, in comparison to group-based methods which are often harder to channel directly into the policy making process (Carnoye and Lopes, 2015). Since money and time represent essential limitations to the development of environmental valuation studies, there is a further need for studies to develop cost-effectiveness analyses of using participatory approaches in the context of ES valuations.

Finally, the use of multi-case valuation studies (Christie et al., 2015; Christie and Rayment, 2012; De Valck et al., 2017; Hanley et al., 2006; Lanz and Provins, 2013; Luisetti et al., 2011; Morrison and Bennett, 2004; Shen et al., 2015) could be advantageous for the generalisability and transferability of environmental valuation study outputs (Stewart, 2012). Nonetheless, this research design could lead to the reduction of the effective sub-sample sizes and potentially decrease the significance levels of the coefficients and thus worsen results robustness. Using cross-validation of the sub-sample estimates could serve as a method to increase the validity of the obtained site-specific WTP estimates. Researchers using multi-case studies might adopt the randomisation process used in the experimental design of this research as it permits to account for the site-specific environmental preferences on the process leading to the final experimental design (Czajkowski, 2016). It is the intention of the researcher to test whether there additional benefits from this ‘site-specific' experimental designs, such as the presence of significant efficiency gains. In other regards, the use of multi-case studies increase the analysis possibilities. For instance, researchers could develop overlay analysis in GIS to identify the most preferred position of restoration projects, using the geo-referenced WTP estimates related to different study sites in addition to other relevant environmental information layers. Finally, multi-case valuation studies can be incorporated into the general practice and could be beneficial for the benefit transfer literature in two ways. First, obtaining a more scattered sample of valuation studies in the UK increase the likelihood of the study and policy location being geographically proximate, which have been found to reduce transfer errors of value transfer (Kaul et al., 2013; Spash and Vatn, 2006). Secondly, the estimates obtained from multiple-case valuation studies can be used to test the accuracy of value transfer exercises for riverine ecosystems (Morrison and Bennett, 2004), green spaces (Perino et al., 2014) and woodland recreation (Bateman et al., 1999).

7.4. Policy implications

Market-based instruments in policies are considered to be flexible and cost-effective tools to resolve environmental issues, such as the degradation and loss of ES (Blackman et al., 2018; C2ES, 2015; Nikolakis and Innes, 2017; Stavins, 2001; Zhang, 2013). However, it has been claimed that the effect of incentive-based approaches aiming to generate environmental progress last as long as the financial intervention persist. While researching this idea some authors have found that the imposition of financial incentives (or disincentives) for more extended periods of time could actually result in the generation of habits or social norms that result in long-term behavioural changes (Goeschl and Perino, 2012; Ho and Yeung, 2015; Kuhfuss et al., 2016; Vollan, 2008; Zhao et al., 2017).

The empirical study developed in the present research provides some insights into the viability of using local taxation (a market-based instrument) to develop an integrative catchment restoration project which improves ES provision levels. The policy proposed in our DCE is in alignment with the restoration project proposed by Natural Scotland (2013b) and therefore is relevant for the application of this supplementary plan of the river basin management. On a more general basis, our analysis can be of interest to policy makers as it provides useful information for guiding the development of environmental plans which aim to restore ES provision levels.

The analysis in this thesis draws together evidence about the influence of external (e.g. sociodemographics), internal (e.g. attitudes) and contextual factors (e.g. local context) in the process of decision making (see chapter 4, 6 and 5, respectively).

The policy implications that arise from our empirical analyses are the following:

i. Environmental management policies which would result in flood control and biodiversity improvements (e.g. natural flood management, green corridors) are more likely to be accepted by Scottish citizens than policies improving the quality of recreational services (see chapter 4, 5 and 6).

ii. The regions with higher environmental quality seem to have more capacity to attract funds to finance ES restoration policies. Thus these areas could be used as a focus of attention to subsidise restoration projects happening in other regions (see chapter 4).

iii. Participating in outdoor recreational activities was found to be associated with higher WTP for estuarine ES improvements and higher pro-environmental attitudes (see chapter 4 and 6). Therefore we recommend environmental management policies which are compatible with the promotion of sustainable outdoor recreational activity (Whiting et al., 2011).

iv. Management plans aiming to restore estuarine ES could benefit from policies which enhance society's environmental consciousness. Our research support previous findings (Dunlap and Heffernan, 1975; Prince, 2017) suggesting that this could be done through the development of local programmes on outdoor recreation education (see chapter 6).

v. The promotion of society’s PEB can be done via the generation of attitudinal changes (see chapter 6). Therefore, the use of economic instruments in policy should thus be complemented with other policies aiming to change environmental behaviour. We suggest environmental education policies as an option, as they tend to generate attitudinal changes through the increase of environmental knowledge (Ajaps and McLellan, 2015; Pooley and O’Connor, 2000).

vi. Finally, policy makers could take advantage of the similarities of the geographical pattern of WTP for different ES to design of spatially explicit policies. For instance, an isoline map based on WTP estimates (or density of local clusters of WTP estimates) could be used to delimitate regions which could be used in differential tax schemes (see chapter 5).

7.5. General discussion and conclusions

Environmental valuation studies can be helpful in providing information which is relevant for the design of public policies to preserve or restore particular ecosystems and thereby sustain the provision of associated ES. The scope for using valuation studies during the design stage of environmental policies depends on each country political, economic and environmental context. However, the UK is considered to be “in the vanguard of those countries actively using environmental valuation in its official policy processes"(Atkinson et al., 2018, p. 2).

The work of Laurans et al. (2013) and Guo and Kildow (2014), systematised the role of ES economic valuation studies in policy making worldwide. To conclude this thesis we will use the categories they developed to discuss how the present research results contribute to the following policy use categories: i) cognitive, ii) operational, and iii) technical.

The first category refers to the use of valuation studies to enhance society's (e.g. general public and decision makers) cognitive understanding of estuarine ES. In this regard, we can argue that our study has not only described how important they are (absolute values), but also explained which estuarine ES are more highly valued (relative values). In addition to this, our study can help to raise the attention of policy makers by demonstrating the economic rationality of investing in programs for ES conservation, since they increase social welfare and represent an essential part of Scottish citizens' well-being. Even though the process determining policy decisions is complex, our results can contribute to the policy discussion by demonstrating that Scottish citizens are willing to fund some of the restoration measures proposed by the river basin management of Natural Scotland (2013).

The second category relates to the operational use of valuation studies and their potential to be integrated into practical decision making processes. Regarding this, the monetary values of estuarine ES generated by this study can be integrated when evaluating the effects of catchment-based policy options while considering the limited budget allocated to ES protection. For instance, policy makers could use the WTP estimates and develop a CBA which compares the use of environmental taxes vs using a PES scheme to develop the restoration measures proposed by the supplementary river basin management project (Natural Scotland, 2013). Within the wide range of policy measures proposed by this project, we found that investing in measures targeting flood control and biodiversity improvements (e.g. natural flood management and green corridors) would increase the social welfare of Scottish citizens. The restoration project could be complemented with local programmes of outdoor recreation education which might raise individuals' willingness to fund the above-mentioned restoration project. The additional costs of developing these policies can be taken into consideration while developing the CBA of policy options.

The last policy use category of environmental valuation studies refers to its technical applications in adjusting economic instruments used for implementing decisions. The value estimates obtained from this study, for instance, are useful when developing or updating the large datasets that technical applications require to simulate socio-environmental systems (e.g. InVEST[18] and SEA[19]). In other regards, our study can be informative in determining the payments to be made by beneficiaries of estuarine ES if the policy aim is to implement catchment-based PES schemes.

In summary, environmental valuation studies can be very informative in guiding the policy process. However, it is important to note that the development of financial incentives should not be considered the *panacea* for promoting environmental changes. In fact, economic instruments could benefit from its integration with other types of policies which provide non-monetary incentives to potentiate their effectiveness and to increase society's WTP for restoring ES. In this sense, results from this research must be considered in integration with other studies approaching the topic from a social and environmental perspective for guiding environmental decision making towards more sustainable estuarine management practices.

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[18] The integrated valuation of environmental services and trade-offs (InVEST) consists of open-source software models which map and value the goods and services society obtain from nature (Natural Capital Project, n.d.).
[19] The strategic environmental assessment (SEA) "is a method of considering and broadly evaluating the likely impact of a public plan, programme or strategy on the environment" (The Scottish Government, 2009, p. 3).

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Annexes

Annex 1 Estuarine ecosystem services within the TEEB framework (Modified from Jacobs et al., 2013)

TEEB Category	Ecosystem service	Benefit	Short description
Provisioning services	Food: Plants	Food	Presence and use of edible plants, including agricultural production for direct food consumption
	Food: Animals	Food	Presence and use of edible animals, including livestock growth and fodder production
	Water for household use	Drinking water	Provision and use of water for household use meeting the quality standards for drinking water
	Water for industrial use	Improved industrial production	Provision and use of water for e.g. cooling water, rinsing water, water for chemical reactions
	Water for agricultural use	Improved agricultural production	Provision and use of water for e.g. irrigation water, freezing prevention for fruit trees, drinking water for cattle
	Water for energy use	Renewable energy production	Provision and use of water for tidal or dam water turbines
	Water for navigation	Shipping	Presence and use of water for shipping purposes
	Raw materials: Renewable soil materials: sand	Building material	Provision and use of sand from dynamic environments which are renewed within a few generations
	Raw materials: Renewable soil materials: clay	Building material	Provision and use of clay from dynamic environments which are renewed within a few generations
	Raw materials: Platform	Building platform for housing, roads, infrastructure	Presence and use of stable and safe environments for building of infrastructure: housing, roads
	Raw materials: Plants	Building material, fibre, fuel	Presence and use of forests, energy and fibre crops
	Raw materials: Animals	Building material, fibre, fuel	Presence and use of animals for fur, leather, gelatine
	Genetic resources	Various improved provisioning services	Presence and use of typical varieties and cultivars of species, adapted to a specific environment
	Medicinal resources	Human health	Presence and use of plants/organisms used in herbal medicine, medicinal tea
	Ornamental resources	Wellbeing	Presence and use of organisms for decorative purposes
Regulating services	Air quality regulation: Removing harmful particles	Human health	Adsorption of fine dust and pollutants on leaf surfaces of forests,

TEEB Category	Ecosystem service	Benefit	Short description
	Air quality regulation: Air-water exchange	Human health	Influence of evaporation and evapotranspiration, condensation on air quality
	Air quality regulation: Biogeochemical reactions due to activity of organisms	Human health	Respiration and photosynthesis, exudation of chemicals by degradation reactions
	Climate regulation: Carbon sequestration and burial	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Buffering carbon stock in living vegetation, burial of organic matter in soils
	Climate regulation: Water thermodynamic regulation	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Cooling effect of vegetation, uptake of solar energy for photosynthesis and evapotranspiration
	Climate regulation: Heat exchange regulation	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Effect of direct reflection, storage, transport, radiation of solar heat by various soil and water bodies
	Regulation extreme events or disturbance: Flood water storage	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Storage of storm or extreme spring tides in natural or flood control habitats
	Regulation extreme events or disturbance: Peak discharge buffering	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Storage of peak discharge floods in natural or flood control habitats
	Regulation extreme events or disturbance: Water current reduction	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Reduction of water current by physical features or vegetation
	Regulation extreme events or disturbance: Wave reduction	Human health, avoided costs caused by extreme events or disturbance, ensured provisioning services	Reduction of wave height by physical features or vegetation
	Regulation extreme events or disturbance: Sound buffering	Human health	Reduction of noise disturbance by presence of natural buffers
	Water quantity regulation: drainage of river water	Ensured platform, food, water, other provisioning services	Drainage of the catchment by the river

TEEB Category	Ecosystem service	Benefit	Short description
	Water quantity regulation: prevention of saline intrusion	Various ensured provisioning services	Countering of saline tidal wave by fresh water discharge
	Water quantity regulation: dissipation of tidal and river energy	Various ensured provisioning services, avoided maintenance costs	Buffering of average flood and discharge variations in the river bed
	Water quantity regulation: landscape maintenance	Various ensured services	Formation and maintenance of typical landscapes and hydrology
	Water quantity regulation: transportation	Shipping	Discharge and tidal input for shipping, including water use for canals and docks
	Water quality regulation: transport of pollutants and excess nutrients	Improved water quality, various ensured services	Transport of pollutants from source, dilution
	Water quality regulation: reduction of excess loads coming from the catchment	Improved water quality, various ensured services	Binding of N, P in sediments and pelagic food web
	Erosion and sedimentation regulation by water bodies	Avoided damage or maintenance costs, various ensured provisioning services	Sediment trapping and gully erosion by variable water currents and topography
	Erosion and sedimentation regulation by biological mediation	Avoided damage or maintenance costs, various ensured provisioning services	Sediment trapping and erosion prevention by vegetation, effects of bioturbation
	Biological regulation of soil processes and soil formation	Various improved provisioning services	Soil microbial activities important for agriculture or water quality regulation processes, bioturbation
	Prevention of establishment of harmful invasive species	Various improved provisioning services	Presence of resilient natural populations able to withstand invasion
	Reduced spread of diseases	Various ensured provisioning services, human health	Presence of resilient and equilibrated natural populations avoiding excessive population growth of disease-carrying vector species, importance for human health or agriculture
	Pollination	Various ensured provisioning services	Presence of pollinators and importance for agricultural production

TEEB Category	Ecosystem service	Benefit	Short description
	Pest control	Insurance of all services	Presence of predators for problematic pest species impacting agricultural production
Habitat or supporting services	Biodiversity	Insurance of all services	Total amount of abiotic and biotic diversity at all levels (gene-landscape), regardless of rarity or vulnerability
Cultural services	Aesthetic information	Wellbeing	Appreciation of beauty of organisms, landscapes
	Opportunities for recreation and tourism	Wellbeing	Opportunities and exploitation for recreation & tourism
	Inspiration for culture, art and design	Wellbeing	Appreciation of organisms, landscapes (inspiration for culture, art and design)
	Spiritual experience	Wellbeing	Appreciation of organisms, landscapes (on a spiritual level)
	Information for cognitive development	Wellbeing	Use of organisms, landscapes for (self-) educational purposes

Annex 2 SAS output for the %mktruns command

SAS Output: The SAS System

Design Summary

Number of Levels	Frequency
3	3
6	1

The SAS System

Saturated = 12

Full Factorial = 162

Some Reasonable Design Sizes	Violations	Cannot Be Divided By
18 *	0	NA
36 *	0	NA

27	4	6 18
12 S	6	9 18
24	6	9 18
30	6	9 18
15	7	6 9 18
2	7	6 9 18
33	7	6 9 18
13	10	3 6 9 18

* - 100% Efficient design can be made with the MktEx macro.

S - Saturated Design - The smallest design that can be made.

The SAS System

nDesignReference				
18	3 ** 6	6 ** 1		Orthogonal Array
36	2 ** 10	3 ** 8	6 ** 1	Orthogonal Array
36	2 ** 9	3 ** 4	6 ** 2	Orthogonal Array
36	2 ** 3	3 ** 9	6 ** 1	Orthogonal Array
36	2 ** 2	3 ** 12	6 ** 1	Orthogonal Array
36	2 ** 2	3 ** 5	6 ** 2	Orthogonal Array
36	2 ** 1	3 ** 8	6 ** 2	Orthogonal Array
36	2 ** 1	3 ** 3	6 ** 3	Orthogonal Array
36	3 ** 7	6 ** 3		Orthogonal Array

Annex 3 Ngene code of pilot experimental design

```
;alts = alt1, alt2,alt3
;rows = 18
;block = 3
;eff = (mnl,d)
;cond:
if (alt1.a=0 and alt1.b=0,alt1.c>0),
if (alt2.a=0 and alt2.b=0,alt2.c>0)
;model:
U(alt1)= a + b1*A[0,1,2]+ b2*B[0,1,2]+ b3*C[0,1,2] + b4*D[5,10,20,50,75,100] /
U(alt2)= a + b1*A[0,1,2]+ b2*B[0,1,2]+ b3*C[0,1,2] + b4*D[5,10,20,50,75,100]
```

Annex 4 Pilot experimental design output

MNL efficiency measures

D error	0.02
A error	0.10
B estimate	100.00
S estimate	0.00

Prior	b1	b2	b3	b4
Fixed prior value	0.00	0.00	0.00	0.00
Sp estimates	Undefined	Undefined	Undefined	Undefined
Sp t-ratios	0.00	0.00	0.00	0.00

Design

Choice situation	alt1.a	alt1.b	alt1.c	alt1.d	alt2.a	alt2.b	alt2.c	alt2.d	Block
1.00	1.00	2.00	2.00	20.00	1.00	0.00	0.00	50.00	3.00
2.00	0.00	1.00	1.00	75.00	2.00	1.00	1.00	10.00	2.00
3.00	1.00	2.00	0.00	50.00	1.00	0.00	2.00	20.00	2.00
4.00	1.00	1.00	2.00	100.00	1.00	1.00	0.00	5.00	3.00
5.00	2.00	1.00	1.00	75.00	0.00	1.00	0.00	10.00	1.00
6.00	0.00	1.00	2.00	10.00	2.00	1.00	0.00	100.00	3.00
7.00	0.00	2.00	0.00	10.00	2.00	0.00	2.00	100.00	2.00
8.00	1.00	1.00	0.00	20.00	2.00	1.00	2.00	50.00	1.00
9.00	2.00	0.00	1.00	10.00	0.00	2.00	1.00	75.00	3.00
10.00	2.00	0.00	0.00	100.00	0.00	2.00	2.00	5.00	3.00

11.00	2.00	1.00	0.00	20.00	1.00	2.00	2.00	50.00	2.00
12.00	1.00	2.00	1.00	50.00	1.00	0.00	1.00	20.00	3.00
13.00	0.00	0.00	1.00	5.00	2.00	2.00	2.00	75.00	2.00
14.00	0.00	2.00	1.00	100.00	2.00	0.00	1.00	5.00	1.00
15.00	2.00	2.00	2.00	5.00	0.00	0.00	1.00	100.00	1.00
16.00	0.00	0.00	2.00	75.00	1.00	2.00	0.00	10.00	1.00
17.00	2.00	2.00	0.00	50.00	0.00	0.00	2.00	20.00	2.00
18.00	2.00	0.00	2.00	5.00	0.00	2.00	0.00	75.00	1.00

Annex 5 Ngene codes of the final experimental design

Clyde code:

```
;alts = alt1*, alt2*,alt3
;rows = 18
;eff = (mnl,d)
;cond:
if (alt1.a=0 and alt1.b=0,alt1.c>0),
if (alt2.a=0 and alt2.b=0,alt2.c>0)
;con
;model:
U(alt1)= b1.dummy[ 2.4642|1.7102 ]*A[2,1,0] + b2.dummy[1.8661|1.1243]*B[2,1,0] +
b3.dummy[ 0.4376|0.6194]*C[2,1,0] + b4[-0.1184]*D[0.5,1.0,2.0,5.0,7.5,10.0] /
U(alt2)= b1.dummy*A[2,1,0] + b2.dummy*B[2,1,0] + b3.dummy*C[2,1,0] +
b4*D[0.5,1.0,2.0,5.0,7.5,10.0]/
U(alt3) = a[-0.3678]$
```

Forth code:

```
;alts = alt1*, alt2*,alt3
;rows = 18
;eff = (mnl,d)
;cond:
if (alt1.a=0 and alt1.b=0,alt1.c>0),
if (alt2.a=0 and alt2.b=0,alt2.c>0)
;con
;model:
U(alt1)= b1.dummy[ 1.2555|1.2513 ]*A[2,1,0] + b2.dummy[1.5367|1.6400]*B[2,1,0] +
b3.dummy[ 0.7991|0.6025]*C[2,1,0] + b4[-0.1201]*D[0.5,1.0,2.0,5.0,7.5,10.0] /
U(alt2)= b1.dummy*A[2,1,0] + b2.dummy*B[2,1,0] + b3.dummy*C[2,1,0] +
b4*D[0.5,1.0,2.0,5.0,7.5,10.0]/
U(alt3) = a[0.3070]$
```

Tay code:

```
;alts = alt1*, alt2*,alt3
;rows = 18
;eff = (mnl,d)
;cond:
if (alt1.a=0 and alt1.b=0,alt1.c>0),
if (alt2.a=0 and alt2.b=0,alt2.c>0)
;con
;model:
```

U(alt1)= b1.dummy[0.9675|0.3178]*A[2,1,0] + b2.dummy[1.7129 |2.1481]*B[2,1,0] + b3.dummy[0.9985|0.9139]*C[2,1,0] + b4[-0.0869]*D[0.5,1.0,2.0,5.0,7.5,10.0] /
U(alt2)= b1.dummy*A[2,1,0] + b2.dummy*B[2,1,0] + b3.dummy*C[2,1,0] + b4*D[0.5,1.0,2.0,5.0,7.5,10.0]/
U(alt3) = a[-0.0329]$

Annex 6 Final site-specific experimental design output

Clyde design:

MNL efficiency measures

D error	0.36
A error	1.25
B estimate	30.50
S estimate	52.23

Prior	b1(d0)	b1(d1)	b2(d0)	b2(d1)	b3(d0)	b3(d1)	b4	a
Fixed prior value	2.46	1.71	1.87	1.12	0.44	0.62	-0.12	-0.37
Sp estimates	1.57	1.92	1.98	3.20	12.04	7.49	3.27	52.23
Sp t-ratios	1.56	1.41	1.39	1.10	0.56	0.72	1.08	0.27

Design

Choice situation	alt1.a	alt1.b	alt1.c	alt1.d	alt2.a	alt2.b	alt2.c	alt2.d
1.00	1.00	1.00	2.00	1.00	2.00	2.00	0.00	7.50
2.00	2.00	1.00	2.00	7.50	1.00	2.00	1.00	1.00
3.00	1.00	2.00	1.00	10.00	2.00	0.00	0.00	0.50
4.00	0.00	0.00	2.00	2.00	0.00	1.00	0.00	7.50
5.00	1.00	0.00	0.00	2.00	0.00	1.00	1.00	2.00
6.00	2.00	0.00	2.00	0.50	1.00	2.00	0.00	10.00
7.00	0.00	2.00	1.00	0.50	2.00	1.00	2.00	10.00
8.00	2.00	0.00	1.00	10.00	0.00	2.00	0.00	0.50
9.00	1.00	1.00	0.00	0.50	2.00	2.00	2.00	10.00
10.00	2.00	2.00	0.00	5.00	1.00	1.00	2.00	1.00
11.00	0.00	1.00	2.00	1.00	1.00	0.00	1.00	5.00
12.00	2.00	1.00	1.00	1.00	1.00	2.00	2.00	5.00
13.00	1.00	0.00	1.00	7.50	0.00	1.00	2.00	0.50
14.00	0.00	1.00	0.00	7.50	0.00	0.00	1.00	1.00
15.00	0.00	1.00	1.00	2.00	2.00	0.00	2.00	5.00
16.00	0.00	2.00	2.00	5.00	1.00	1.00	0.00	2.00
17.00	0.00	0.00	1.00	10.00	0.00	0.00	2.00	7.50
18.00	1.00	2.00	2.00	5.00	2.00	0.00	1.00	2.00

Forth design:

MNL efficiency measures

D error	0.31
A error	0.97
B estimate	48.35
S estimate	69.89

Prior	b1(d0)	b1(d1)	b2(d0)	b2(d1)	b3(d0)	b3(d1)	b4	a
Fixed prior value	1.26	1.25	1.54	1.64	0.80	0.60	-0.12	0.31
Sp estimates	2.50	2.47	2.03	1.87	4.52	7.06	2.76	69.89
Sp t-ratios	1.24	1.25	1.38	1.43	0.92	0.74	1.18	0.23

Design

Choice situation	alt1.a	alt1.b	alt1.c	alt1.d	alt2.a	alt2.b	alt2.c	alt2.d
1.00	1.00	2.00	1.00	5.00	0.00	1.00	2.00	1.00
2.00	0.00	2.00	0.00	0.50	1.00	0.00	1.00	7.50
3.00	2.00	0.00	2.00	2.00	0.00	1.00	0.00	5.00
4.00	2.00	0.00	1.00	0.50	1.00	2.00	0.00	10.00
5.00	2.00	1.00	0.00	10.00	1.00	0.00	2.00	0.50
6.00	2.00	0.00	0.00	1.00	0.00	2.00	2.00	7.50
7.00	0.00	2.00	2.00	1.00	1.00	1.00	1.00	5.00
8.00	1.00	2.00	2.00	10.00	0.00	1.00	1.00	0.50
9.00	2.00	1.00	2.00	5.00	1.00	2.00	1.00	1.00
10.00	1.00	2.00	0.00	1.00	2.00	1.00	2.00	7.50
11.00	2.00	2.00	2.00	7.50	1.00	1.00	0.00	1.00
12.00	0.00	1.00	1.00	5.00	2.00	0.00	0.00	2.00
13.00	1.00	1.00	1.00	7.50	2.00	2.00	0.00	0.50
14.00	0.00	1.00	2.00	2.00	2.00	0.00	1.00	5.00
15.00	0.00	0.00	1.00	7.50	0.00	0.00	2.00	10.00
16.00	1.00	0.00	0.00	10.00	0.00	0.00	1.00	2.00
17.00	1.00	0.00	2.00	0.50	2.00	2.00	1.00	10.00
18.00	0.00	2.00	1.00	2.00	2.00	1.00	0.00	2.00

Tay design:

MNL efficiency measures

D error	0.33
A error	1.04
B estimate	41.72
S estimate	5895.51

Prior	b1(d0)	b1(d1)	b2(d0)	b2(d1)	b3(d0)	b3(d1)	b4	a
Fixed prior value	0.97	0.32	1.71	2.15	1.00	0.91	-0.09	-0.03
Sp estimates	3.61	22.46	1.98	1.70	3.27	3.53	4.12	5895.51
Sp t-ratios	1.03	0.41	1.39	1.50	1.08	1.04	0.97	0.03

Design

Choice situation	alt1.a	alt1.b	alt1.c	alt1.d	alt2.a	alt2.b	alt2.c	alt2.d
1.00	1.00	0.00	1.00	0.50	0.00	1.00	0.00	10.00
2.00	0.00	0.00	2.00	1.00	2.00	0.00	0.00	7.50
3.00	0.00	1.00	0.00	2.00	1.00	2.00	1.00	5.00
4.00	1.00	1.00	1.00	10.00	2.00	2.00	0.00	0.50
5.00	1.00	2.00	0.00	1.00	2.00	0.00	2.00	5.00
6.00	2.00	2.00	2.00	5.00	0.00	1.00	1.00	2.00
7.00	2.00	1.00	0.00	7.50	0.00	2.00	2.00	1.00
8.00	2.00	2.00	0.00	5.00	1.00	1.00	1.00	1.00
9.00	2.00	0.00	0.00	1.00	1.00	0.00	1.00	7.50
10.00	1.00	1.00	0.00	7.50	0.00	2.00	1.00	1.00
11.00	0.00	0.00	1.00	10.00	1.00	0.00	0.00	2.00
12.00	1.00	0.00	0.00	2.00	0.00	0.00	2.00	10.00
13.00	1.00	2.00	2.00	7.50	2.00	1.00	0.00	0.50
14.00	2.00	0.00	2.00	5.00	0.00	2.00	0.00	2.00
15.00	2.00	0.00	1.00	0.50	1.00	2.00	2.00	10.00
16.00	0.00	1.00	1.00	10.00	1.00	0.00	2.00	0.50
17.00	0.00	1.00	2.00	0.50	2.00	2.00	1.00	7.50
18.00	0.00	2.00	1.00	2.00	2.00	1.00	2.00	5.00

Annex 7 Survey ethical approval

University of St Andrews

University Teaching and Research Ethics Committee
School Of Geography And Geosciences

3rd March 2016
Valeria Maria Toledo-Gallegos
Geography and Geosciences

Ethics Reference No: *Please quote this ref on all correspondence*	GG11966
Project Title:	Mapping ecosystem services and value changes in Scottish Estuaries
Researchers Name(s):	Valeria Maria Toledo-Gallegos
Supervisor(s):	Professor Nick Hanley

Thank you for submitting your application which was considered by the Geography and Geosciences School Ethics Committee on the date specified below. The following documents were reviewed:

1. Ethical Application Form 29th February 2016

The University Teaching and Research Ethics Committee (UTREC) approves this study from an ethical point of view. Please note that where approval is given by a School Ethics Committee that committee is part of UTREC and is delegated to act for UTREC.

Approval is given for three years. Projects, which have not commenced within two years of original approval, must be re-submitted to your School Ethics Committee.

You must inform your School Ethics Committee when the research has been completed. If you are unable to complete your research within the 3 three year validation period, you will be required to write to your School Ethics Committee and to UTREC (where approval was given by UTREC) to request an extension or you will need to re-apply.

Any serious adverse events or significant change which occurs in connection with this study and/or which may alter its ethical consideration, must be reported immediately to the School Ethics Committee, and an Ethical Amendment Form submitted where appropriate.

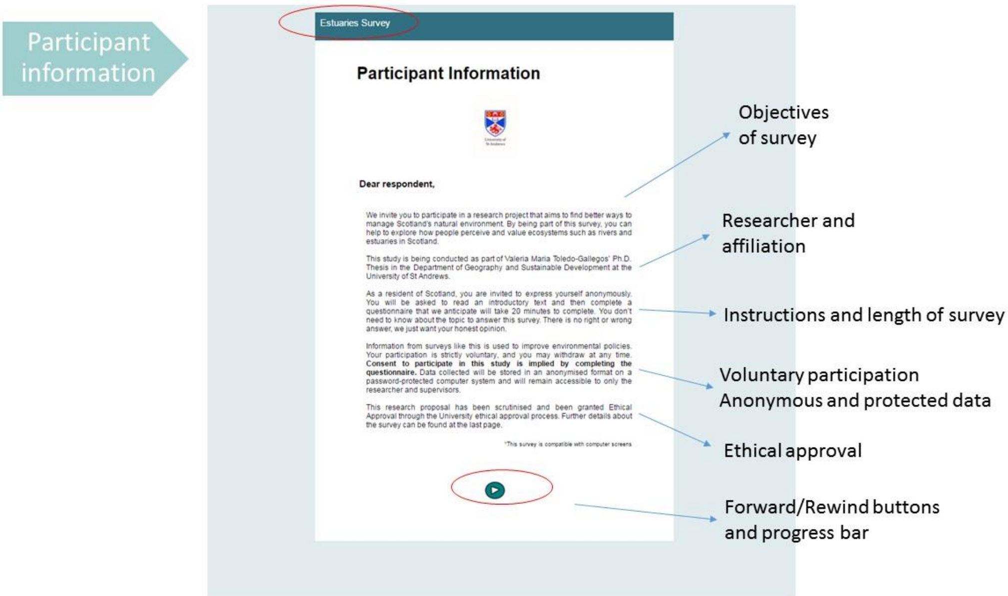
Approval is given on the understanding that the 'Guidelines for Ethical Research Practice' (http://www.st-andrews.ac.uk/media/UTRECguidelines%20Feb%2008.pdf) are adhered to.

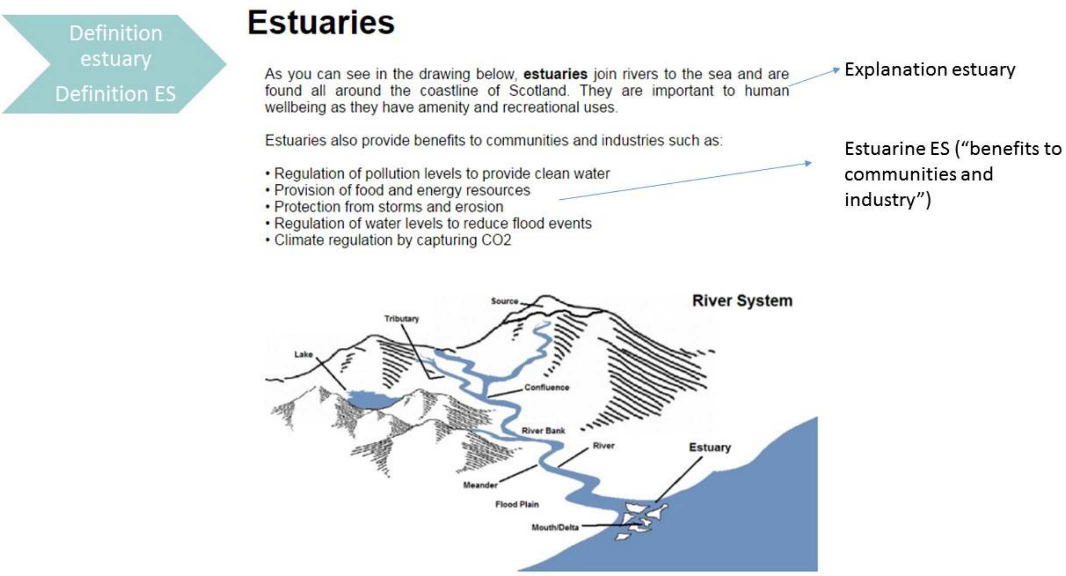
Yours sincerely,

Dr. Matt Southern
Convenor of the School Ethics Committee

UTREC School of Geography and Geosciences Convenor, Irvine Building, North Street, St Andrews, KY16 9AL
Email: ggethics@st-andrews.ac.uk Tel: 01334 463897
The University of St Andrews is a charity registered in Scotland: No SC013532

Annex 8 Full-length questionnaire explanation





Definition study area

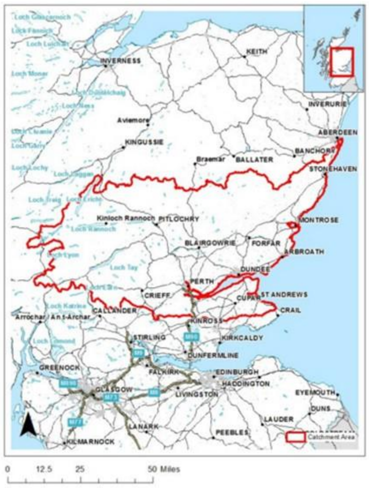
Residency

The Tay estuary

The **Tay estuary** is located at the mouth of the Tay River. For this study, we consider the area of land which collects water heading to the Tay estuary. This area is delineated by a thick line on the following map:

Definition of catchment area
Delineation of interest area

Tay Area



Do you live inside the Tay area? (Inside the thick line)

- Yes
- No

- To identify WTP differences among residents and not residents
- Warm up question

Policy explanation

Policy objectives

Restoration project

How could we restore the Tay area?

Environmental impacts

The Tay area has been changing over time due to urban development and rural land management. To maintain the wide range of benefits we get from the Tay estuary, it may be necessary to improve the physical condition of rivers and estuaries in the area with a "restoration project". **Restoration projects** vary in accordance with local necessities, but they are measures aiming to reverse historical damage to ecosystems.

Restoration project objective: maintain ES

Explanation of restoration projects

Useful for decision-makers

This questionnaire aims to inform decision-makers by exploring your preferences for managing the Tay area. Because environmental policies are costly and the governmental budget for developing them is limited, we restricted this study to the following benefits we get from estuaries:

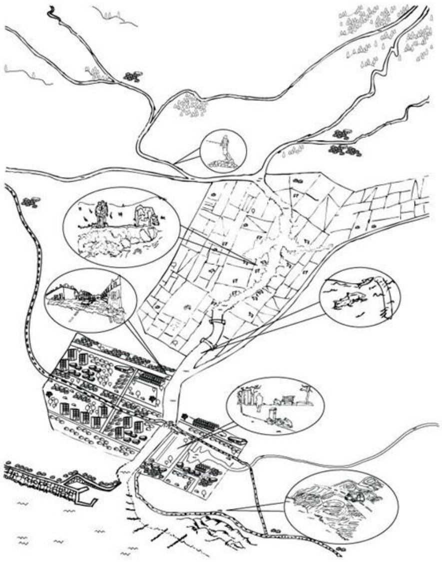
Justifying ES of interest

- Flood control
- Biodiversity
- Recreation

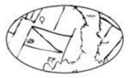
The following drawings explain some of the possible restoration measures that could improve the conditions of river and estuaries. On this page, we show a **"before policy"** scenario, which explains current problems affecting our wellbeing.

On the next page you will see the **"after policy"** scenario, which explains how some of these problems can be improved. Some restoration measures improve all type of benefits we get from the environment (flood control, biodiversity and recreation), whereas others may only affect one of them (such as flood control).

Tay river before restoration project



In the drawing you can see:



Lack of vegetation upstream, and on the banks of rivers causes erosion.

Damages to rivers and artificial barriers stop fish migration, meaning that fishing is poor.

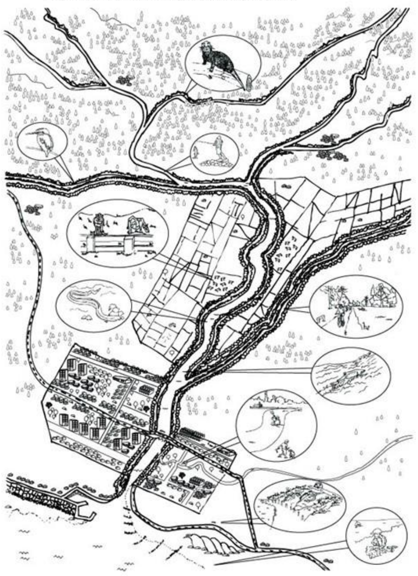
Cities and farming are at the edge of the river and streams, damaging their ecological health and biodiversity.

People have limited access to the riverside and shoreline, as there is a lack of paths and recreation areas.

Rivers have been straightened and drained. Water causes erosion and flooding during periods of heavy rain.

Unstable sand dunes lead to poor wildlife habitat and fewer recreational opportunities at the beach.

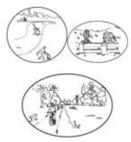
Tay river after restoration project



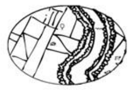
In the drawing you can see:

Improvements in vegetation located upstream (woodland), in the riverbank (wetland) and coast (saltmarsh), reduces erosion while improving wildlife habitats.

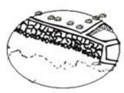
Removing old weirs and dams in rivers allow fish migration.



Creating buffer strips alongside rivers and fencing the cattle reduces environmental damage and increases biodiversity while providing more space for recreational activities.



Naturalizing the shape of rivers and restoring vegetation reduces water speed and so reduces downstream flooding.



Coastal flooding is reduced by natural flood defences such as saltmarshes.

Sand dunes have been stabilized, creating wildlife habitats and improving recreational opportunities at the beach.

Recreational use questions

First, we would like to ask about your outdoor recreational use of the area.

Tay Area

Have you visited the Tay area for leisure and outdoor recreational activities in the last 12 months?

- Yes
- No

Thinking about the **last 12 months**, how often have you done each activity in the Tay area?

	Every day (once or more)	Once or several times a week	Once or twice a month	Once every 2-3 months	Once or twice	Never
Walking, running or jogging						
Hill walking or climbing						
Cycling or mountain biking						
Wildlife\nature watching						
Sightseeing or visiting attractions\historic sites						
Picnicking						
Recreational fishing						
Swimming outdoors						
Water Sports						
Beach Sports						
Snow Sports						
Golf						
Motor Cruising						
Wild camping						
Horse riding						

Water sports: canoeing, kayaking, surfing, windsurfing, kite surfing, paddle boarding, windsurfing, rowing, sculling, yachting and sailing, SCUBA diving, snorkelling.
Beach sports: land yachting, power kiting, kite bugging.
Motor Cruising: jet skis, motor boat, ship.
Snow sports: skiing, snowboarding.

Thinking about your **last visit**, please mention the name of the MAIN destination in the Tay area?

- To identify users and non-users of recreational service
- To identify frequency and type of use
- To identify destinations and length of travel time (with postcode information)

Flood perception

Now, we would like to ask some questions about flooding.

Please indicate how well you agree with the following statements:

	Strongly agree	Agree	Neutral	Disagree	Strongly disagree	I don't know
I am concerned about flooding						
The frequency and extent of flooding are increasing where I live						
I am worried that the current flood defences are not adequate enough to protect my home						

- To explore flood perception and awareness
- To explore perception of flood defences

Flood risk is commonly measured by the chance for a flood event to occur. What do you think are the chances of your home being flooded?

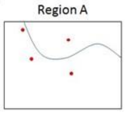
- High (once in every ten years or 10% chance of happening in any one year)
- Medium (once in every fifty years or 2% chance of happening in any one year)
- Low (once in every hundred years or 1% chance of happening in any one year)
- No flood risk (it has never happened or less than 1% chance of happening in any one year)
- I don't know

How many years have you lived in your current home?

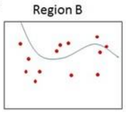
- To get a perceived measure of flood risk (perception of flood influences WTP)
- To see if "I don't know" answers are linked with residency time at home

Imagine two identical regions nearby a river. If each red dot in the figures represents all flood events registered in one year, which of the following regions has a higher risk of flooding?

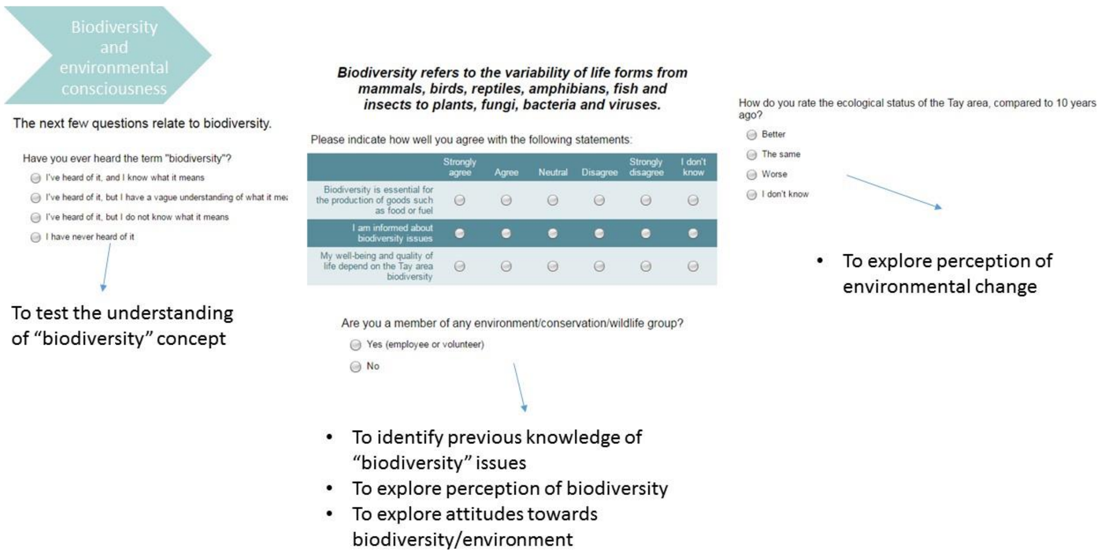
Region A



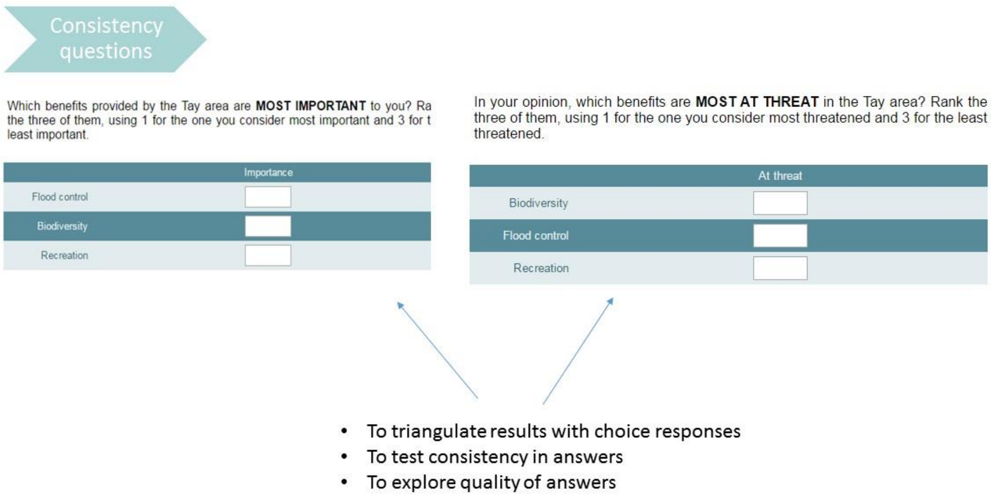
Region B



- To test the understanding of "risk" concept
- To indicate the quality of responses



Biodiversity and environmental consciousness
The next few questions relate to biodiversity.
Have you ever heard the term "biodiversity"?
I've heard of it, and I know what it means
I've heard of it, but I have a vague understanding of what it mea
I've heard of it, but I do not know what it means
I have never heard of it
To test the understanding of "biodiversity" concept
Biodiversity refers to the variability of life forms from mammals, birds, reptiles, amphibians, fish and insects to plants, fungi, bacteria and viruses.
Please indicate how well you agree with the following statements:
Strongly agree
Agree
Neutral
Disagree
Strongly disagree
I don't know
Biodiversity is essential for the production of goods such as food or fuel
I am informed about biodiversity issues
My well-being and quality of life depend on the Tay area biodiversity
Are you a member of any environment/conservation/wildlife group?
Yes (employee or volunteer)
No
• To identify previous knowledge of "biodiversity" issues
• To explore perception of biodiversity
• To explore attitudes towards biodiversity/environment
How do you rate the ecological status of the Tay area, compared to 10 years ago?
Better
The same
Worse
I don't know
• To explore perception of environmental change



Consistency questions
Which benefits provided by the Tay area are MOST IMPORTANT to you? Ra the three of them, using 1 for the one you consider most important and 3 for t least important.
Importance
Flood control
Biodiversity
Recreation
In your opinion, which benefits are MOST AT THREAT in the Tay area? Rank the three of them, using 1 for the one you consider most threatened and 3 for the least threatened.
At threat
Biodiversity
Flood control
Recreation
• To triangulate results with choice responses
• To test consistency in answers
• To explore quality of answers

Attribute explanation

The following section explores your preferences for developing a restoration project in the Tay area. This would increase the benefits provided by the Tay estuary.

Area where the restoration project would be implemented

To go ahead, the restoration project would need to be funded through an **increase in your future annual tax payments** (lasting 10 years).

Payment vehicle and time frame

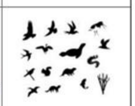
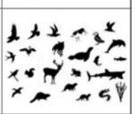
The kind of measures taken to restore the Tay area will determine the way in which it provides benefits to society such as flood control, biodiversity and recreation.

On the following pages, we explain **three possible levels** that each of these benefits could take in the future:

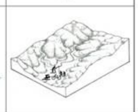
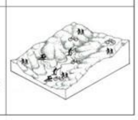
Flood control

Increase in flood risk: The frequency of flood events keeps increasing in the area (more events each decade and more chances each year). Flood defences fail because straightening rivers and the absence of vegetation keeps a high-speed flow of water. The failure to provide a free space between the river and human activities (buffers) also mean that the extent of residential and agricultural damages keeps increasing in time.	
Slight reduction in flood risk: Flooding occurs every fifty years in the area. Already installed flood defences are useful because the restoration of the curvy shape of rivers helps to reduce speed flow. The extent of residential and agricultural damages is reduced significantly as buffers are created in some areas.	
Large reduction in flood risk: Flooding occurs every hundred years in the area. No need for new flood defences as the restoration of the river shape and vegetation (upstream, in river plain and riverside) helped to lower speed flow. Residential and agricultural damages have almost completely been avoided with the creation of buffers.	

Biodiversity

Decrease in biodiversity: The chances of observing any type of wildlife (fish, birds, butterflies, mammals or reptiles) are reduced in the area because habitat degradation continues. Endangered species disappear.	
Slight increase in biodiversity: Improvement in chances of observing birds, butterflies and few mammals happen when restoring ecosystems with native vegetation in the area. An increase in the observable number of endangered species happens inside protected areas.	
Large increase in biodiversity: Improvement in chances of observing fish, birds, butterflies, mammals and reptiles happen when restoring ecosystems and eliminating structures that act as barriers to wildlife movement. An increase in the observable number of endangered species happens inside and outside protected areas.	

Recreation

Decrease in recreation: The quality of outdoor recreation decreases. Degradation of nature has led to non-scenic areas. Insufficient and not well-maintained infrastructure hinders access to the riverside and shoreline. Wildlife watching is possible everywhere but no walking, cycling, recreational fishing, swimming and other water sports.	
Slight increase in recreation: Restoration and greening policies have increased the scenic quality and access. A path network with multi-purpose trails and resting places has been developed in few isolated areas, improving its quality of outdoor recreation. Wildlife watching, walking, cycling, recreational fishing, swimming and other water sports is possible ONLY in those areas.	
Large increase in recreation: Restoration and greening policies have increased the scenic quality and access. A path network connects woodland, cities and coast with multi-purpose trails and resting places. Wildlife watching, walking, cycling, recreational fishing, swimming and other water sports can be developed all around the area. The quality of outdoor recreation increases everywhere.	

- To define what is meant by “increase”, “decrease”, “slight” and “large”
- To exemplify possible and feasible future scenarios

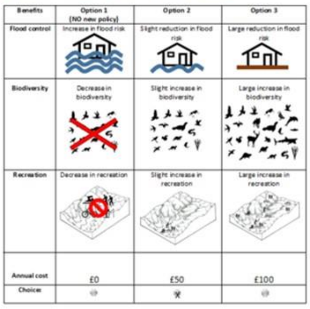
Choice card explanation

Now, you will be presented with six different choice cards.

Each one contains three possible management options taking place in the Tay area and with different potential for providing benefits to society. All options are composed of the combination of 4 characteristics:

- Flood control
- Biodiversity
- Recreation
- Cost of the policy (increase in your tax bill)

Here is an example of what a choice card looks like:



Benefits	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Slight reduction in flood risk	Large reduction in flood risk
Biodiversity	Decrease in biodiversity	Slight increase in biodiversity	Large increase in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Large increase in recreation
Annual cost	£0	£50	£100
Choice:			

Instructions for understanding choice cards

If no actions are taken, degradation of the Tay area will lead to the reduction of benefits provided by estuaries in time. Option 1 reflects a "worsening" scenario, happening in the next 10 years if NO new policy is created and NO new measures are taken for protecting the estuary. Alternatively, if restoration measures are taken benefits provided by estuaries will increase in time. Option 2 and 3 represent "improvement" scenarios happening in the next 10 years if a new policy has been applied.

On each choice card, you will be asked to tick ONE management option which should be the one you would most want to go ahead. Remember to think about the increase in your annual tax payment and what this would mean for your household. Only select the options you can afford and you are willing to pay.

Text to emphasize on:

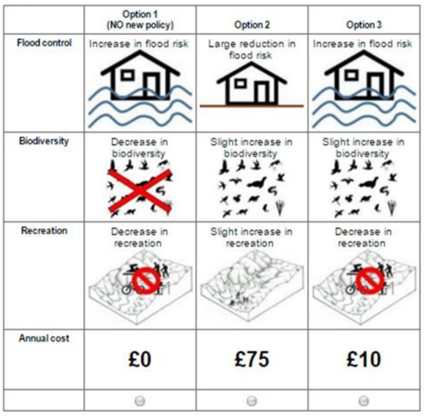
- The existence of a "good" and "bad" scenario
- The time frame of environmental changes
- The relevance of the cost attribute

Choice tasks

6 choice tasks each person

If these were your only options, which would you choose?

(1 of 6)



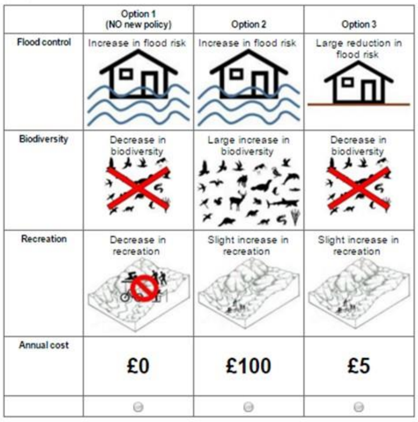
	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Large reduction in flood risk	Increase in flood risk
Biodiversity	Decrease in biodiversity	Slight increase in biodiversity	Slight increase in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Decrease in recreation
Annual cost	£0	£75	£10

Worsening scenario

Improvement scenario

If these were your only options, which would you choose?

(3 of 6)



	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Increase in flood risk	Large reduction in flood risk
Biodiversity	Decrease in biodiversity	Large increase in biodiversity	Decrease in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Slight increase in recreation
Annual cost	£0	£100	£5

If these were your only options, which would you choose?
(1 of 6)

	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Large reduction in flood risk	Increase in flood risk
Biodiversity	Decrease in biodiversity	Slight increase in biodiversity	Slight increase in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Decrease in recreation
Annual cost	£0	£75	£10

If these were your only options, which would you choose?
(3 of 6)

	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Increase in flood risk	Large reduction in flood risk
Biodiversity	Decrease in biodiversity	Large increase in biodiversity	Decrease in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Slight increase in recreation
Annual cost	£0	£100	£5

If these were your only options, which would you choose?
(1 of 6)

	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Large reduction in flood risk	Increase in flood risk
Biodiversity	Decrease in biodiversity	Slight increase in biodiversity	Slight increase in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Decrease in recreation
Annual cost	£0	£75	£10

If these were your only options, which would you choose?
(3 of 6)

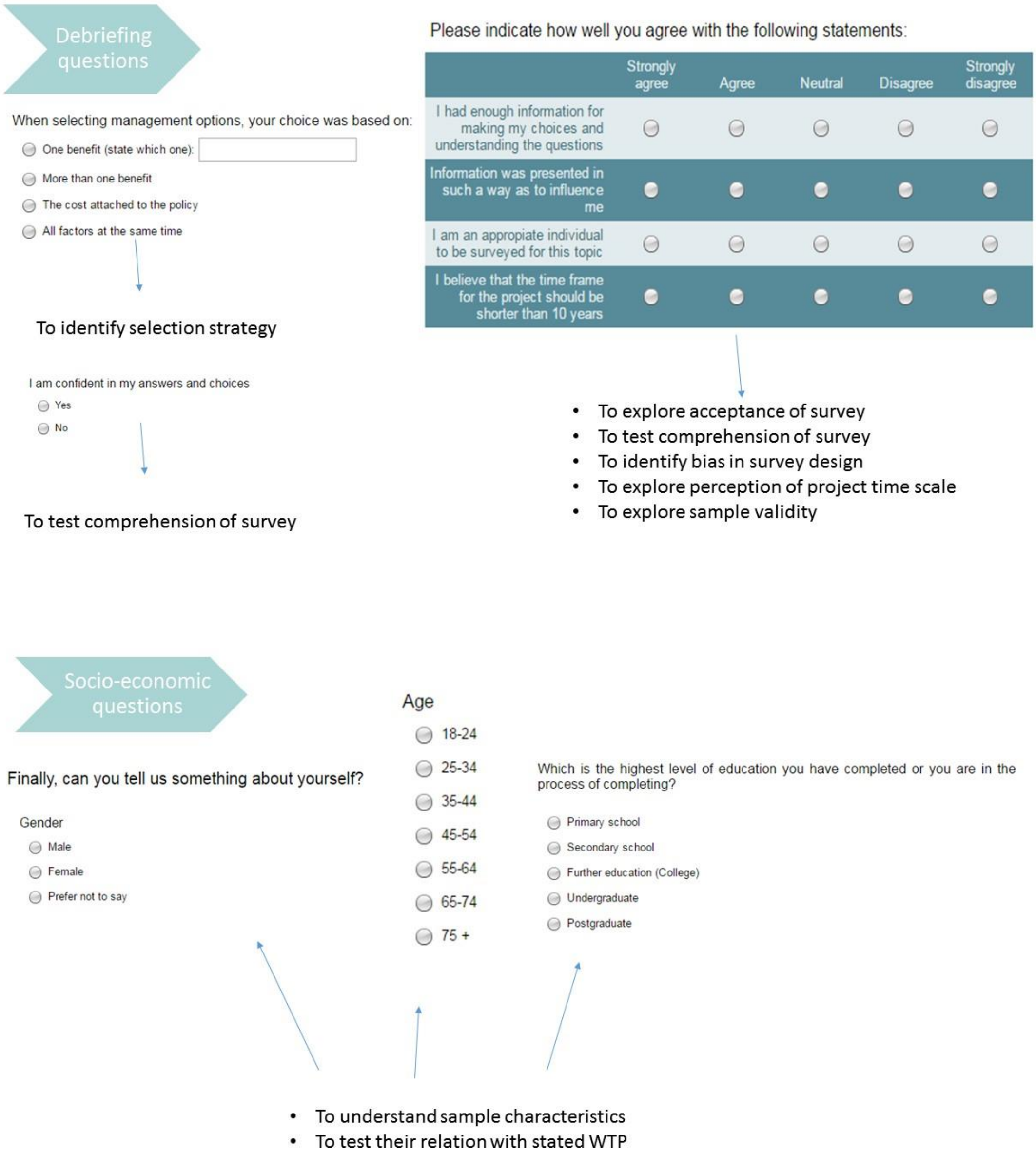
	Option 1 (NO new policy)	Option 2	Option 3
Flood control	Increase in flood risk	Increase in flood risk	Large reduction in flood risk
Biodiversity	Decrease in biodiversity	Large increase in biodiversity	Decrease in biodiversity
Recreation	Decrease in recreation	Slight increase in recreation	Slight increase in recreation
Annual cost	£0	£100	£5

Zero bid question

- To identify reasons for eliciting zero WTP
- To identify protest answers
- To identify tax aversion and attitudes influencing responses
- To explore the quality of the survey

Why did you choose the "NO new policy" option in one or more choice cards? Select the statement that best describe your reason.

- I cannot afford to pay
- I don't believe that my payment will be used effectively
- I believe I should not be the one paying for it
- I don't believe there is a need for a restoration project and priorities for public funds should be different
- I don't think the suggested policies will improve benefits provided by estuaries
- I don't think the suggested policies are viable
- I don't pay taxes and/or I would prefer another mechanism for paying
- Other


Debriefing questions
When selecting management options, your choice was based on:
One benefit (state which one):
More than one benefit
The cost attached to the policy
All factors at the same time
To identify selection strategy
I am confident in my answers and choices
Yes
No
To test comprehension of survey
Please indicate how well you agree with the following statements:
Strongly agree
Agree
Neutral
Disagree
Strongly disagree
I had enough information for making my choices and understanding the questions
Information was presented in such a way as to influence me
I am an appropiate individual to be surveyed for this topic
I believe that the time frame for the project should be shorter than 10 years
To explore acceptance of survey
To test comprehension of survey
To identify bias in survey design
To explore perception of project time scale
To explore sample validity
Socio-economic questions
Finally, can you tell us something about yourself?
Gender
Male
Female
Prefer not to say
Age
18-24
25-34
35-44
45-54
55-64
65-74
75 +
Which is the highest level of education you have completed or you are in the process of completing?
Primary school
Secondary school
Further education (College)
Undergraduate
Postgraduate
To understand sample characteristics
To test their relation with stated WTP

Which is the postcode of your residence?

Important for spatial analysis:
Non compulsory
Min 4 digits
Max 8 digits

- To explore spatial distribution of WTP
- To locate individuals inside catchment areas
- To get a spatial reference for information on perceived flood risk

How many people are there in your household, including any children or/and babies?

Which member of your household is the Chief Income Earner (person with the largest income)?

- Respondent
- Respondent's spouse/partner
- Other adult

Working status of Chief Income Earner

- Employed
- Self-employed
- Not working, dependent on state benefit
- Not working, other income

Which group represents the Chief Income Earner's total income before deductions (tax, National Insurance etc.)?

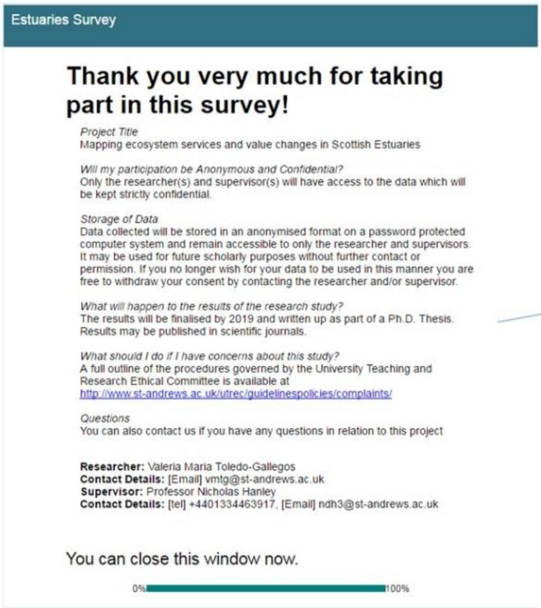
Weekly	Monthly	Annual
Up to £49	Up to £216	Up to £2,599
£50 up to £99	£217 up to £432	£2,600 up to £5,199
£100 up to £199	£433 up to £866	£5,200 up to £10,399
£200 up to £299	£867 up to £1,299	£10,400 up to £15,599
£300 up to £399	£1,300 up to £1,732	£15,600 up to £20,799
£400 up to £499	£1,733 up to £2,166	£20,800 up to £25,999
£500 up to £599	£2,167 up to £2,599	£26,000 up to £31,199
£600 up to £699	£2,600 up to £3,032	£31,200 up to £36,399
£700 up to £799	£3,033 up to £3,466	£36,400 up to £41,599
£800 up to £899	£3,467 up to £3,899	£41,600 up to £46,799
£900 up to £999	£3,900 up to £4,332	£46,800 up to £51,999
£1000 or more	£4,333 or more	£52,000 or more

- Prefer not to say

- To understand household characteristics
- To test validity of WTP responses
- To have useful information for WTP aggregation

You have reached the end of this survey. Please note that once the "forward" button is clicked, your responses will be submitted to us.

Closing information



Estuaries Survey

Thank you very much for taking part in this survey!

Project Title
Mapping ecosystem services and value changes in Scottish Estuaries

Will my participation be Anonymous and Confidential?
Only the researcher(s) and supervisor(s) will have access to the data which will be kept strictly confidential.

Storage of Data
Data collected will be stored in an anonymised format on a password protected computer system and remain accessible to only the researcher and supervisors. It may be used for future scholarly purposes without further contact or permission. If you no longer wish for your data to be used in this manner you are free to withdraw your consent by contacting the researcher and/or supervisor.

What will happen to the results of the research study?
The results will be finalised by 2019 and written up as part of a Ph.D. Thesis. Results may be published in scientific journals.

What should I do if I have concerns about this study?
A full outline of the procedures governed by the University Teaching and Research Ethical Committee is available at
http://www.st-andrews.ac.uk/utrec/guidelinespolicies/complaints/

Questions
You can also contact us if you have any questions in relation to this project

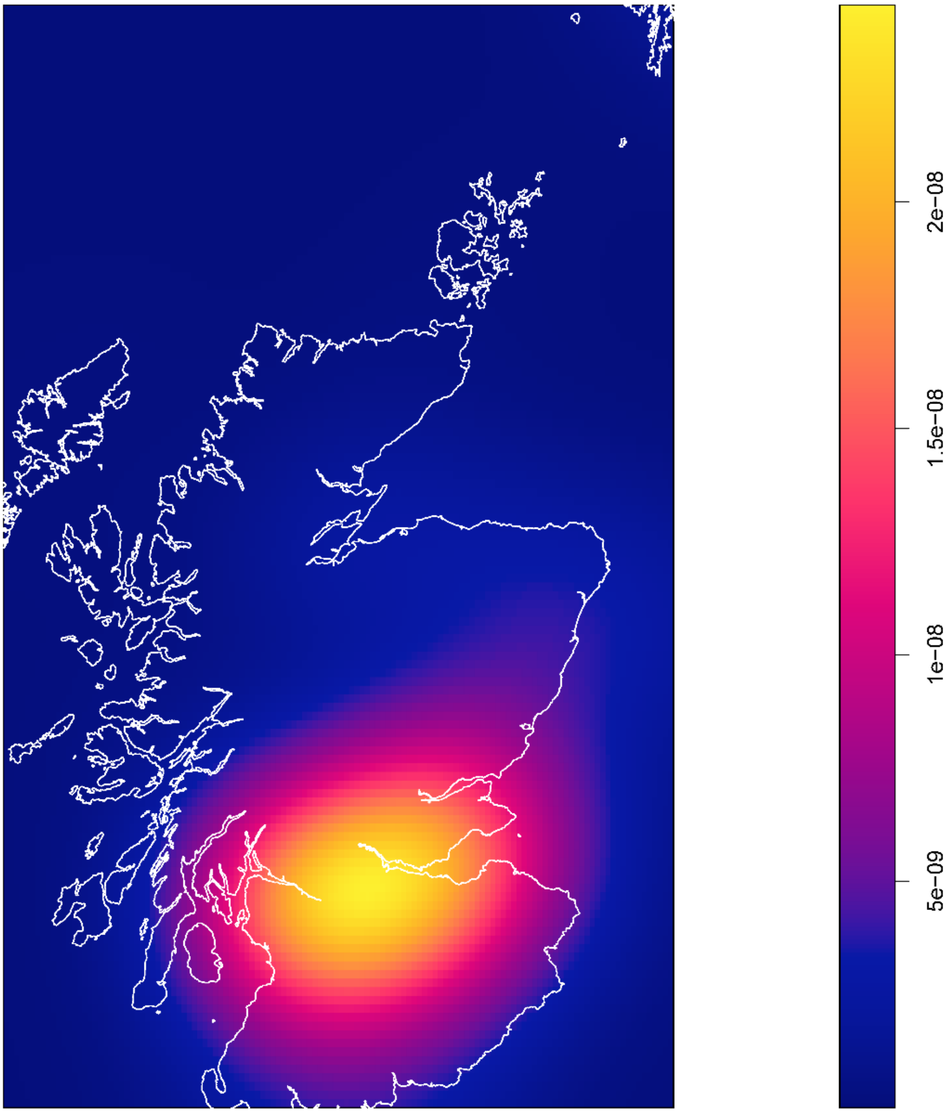
Researcher: Valeria Maria Toledo-Gallegos
Contact Details: [Email] vmtg@st-andrews.ac.uk
Supervisor: Professor Nicholas Hanley
Contact Details: [tel] +4401334463917, [Email] ndh3@st-andrews.ac.uk

You can close this window now.

0% 100%

Closure page with thanks and additional information

Annex 9 Survey point density



Annex 10 Re-specified attribute interacted models

		RPL interacted 5					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Resident	All	-0.73	***	0.20	-0.06	***	0.28
	Clyde	-1.52	***	0.37	-0.26	***	0.47
	Forth	-0.28	***	0.35	0.24	***	0.54
	Tay	-0.04	***	0.60	-1.24	***	0.75
F2*Resident	All	-1.22	***	0.24	-0.08	***	0.25
	Clyde	-2.22	***	0.46	-0.87	***	0.44
	Forth	-0.90	***	0.44	-0.37	***	0.47
	Tay	0.31	***	0.76	-3.08	***	0.90
B1*Resident	All	-0.40	***	0.22	0.06	***	0.38
	Clyde	-1.08	***	0.38	-0.13	***	0.75
	Forth	-0.21	***	0.39	0.41	***	0.72
	Tay	0.49	***	0.60	0.16	***	0.92
B2*Resident	All	-0.61	***	0.23	0.65	***	0.40
	Clyde	-1.28	***	0.40	-1.06	***	0.58
	Forth	-0.35	***	0.43	-0.16	***	0.47
	Tay	1.14	***	0.71	-1.07	***	0.80
R1*Resident	All	0.11	***	0.16	0.20	***	0.43
	Clyde	-0.14	***	0.27	-0.06	***	0.70
	Forth	0.32	***	0.28	-0.05	***	0.72
	Tay	-0.26	***	0.46	-0.92	***	0.85
R2*Resident	All	0.46	***	0.17	0.11	***	0.27
	Clyde	0.59	***	0.28	-0.29	***	0.64
	Forth	0.45	***	0.31	0.27	***	0.41
	Tay	0.68	***	0.46	-0.79	***	0.84
Cost*Resident	All	0.00	***	0.00	-	-	-
	Clyde	0.01	***	0.00	-	-	-
	Forth	0.01	***	0.00	-	-	-
	Tay	-0.01	***	0.01	-	-	-
F1	All	1.92	***	0.13	-0.63	***	0.17
	Clyde	2.52	***	0.31	0.82	***	0.35
	Forth	1.85	***	0.24	-0.78	***	0.30
	Tay	1.62	***	0.17	-0.43	***	0.28
F2	All	2.54	***	0.16	-1.11	***	0.15
	Clyde	3.40	***	0.41	1.73	***	0.35
	Forth	2.28	***	0.29	-1.16	***	0.29
	Tay	2.27	***	0.22	0.71	***	0.21
B1	All	1.83	***	0.14	0.34	***	0.24
	Clyde	2.45	***	0.31	0.35	***	0.49
	Forth	1.76	***	0.26	-0.71	***	0.30
	Tay	1.50	***	0.19	0.00	***	0.32
B2	All	2.04	***	0.16	-0.99	***	0.14
	Clyde	2.35	***	0.33	1.09	***	0.30
	Forth	2.04	***	0.29	0.96	***	0.28
	Tay	1.83	***	0.22	0.91	***	0.19
R1	All	0.61	***	0.09	0.01	***	0.21
	Clyde	0.76	***	0.21	0.05	***	0.50
	Forth	0.66	***	0.18	0.02	***	0.26
	Tay	0.49	***	0.13	0.01	***	0.49
R2	All	0.50	***	0.10	0.53	***	0.17
	Clyde	0.42	***	0.22	0.59	***	0.42
	Forth	0.44	***	0.18	0.71	***	0.27

		RPL interacted 5					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Tay	0.53	***	0.13	-0.17	***	0.49
Cost	All	-0.02	***	0.00	-	-	-
	Clyde	-0.03	***	0.00	-	-	-
	Forth	-0.02	***	0.00	-	-	-
	Tay	-0.01	***	0.00	-	-	-
ASC	All	-1.84	***	0.29	3.31	***	0.27
	Clyde	-1.80	***	0.54	3.79	***	0.52
	Forth	-1.83	***	0.51	3.36	***	0.48
	Tay	-1.87	***	0.46	2.77	***	0.39
Log-likelihood	All	-2727.14					
	Clyde	-1278.79					
	Forth	-1298.56					
	Tay	1188.00					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.30					
	Forth	0.27					
	Tay	0.29					
AIC	All	5510.27					
	Clyde	1800.55					
	Forth	1888.85					
	Tay	1857.33					
BIC	All	5683.04					
	Clyde	1942.22					
	Forth	2030.95					
	Tay	1999.57					

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

	RPL interacted 6						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Visitor	All	-0.18	***	0.19	-0.19	***	0.28
	Clyde	-0.85	***	0.36	0.33	***	0.62
	Forth	0.20	***	0.34	-0.01	***	0.47
	Tay	0.26	***	0.33	-0.87	***	0.75
F2*Visitor	All	-0.29	***	0.23	-0.21	***	0.24
	Clyde	-1.06	***	0.45	-0.26	***	0.43
	Forth	0.03	***	0.41	2.44	***	0.50
	Tay	0.30	***	0.39	0.11	***	0.43
B1*Visitor	All	-0.46	***	0.21	-0.33	***	0.53
	Clyde	-0.85	***	0.40	-0.58	***	0.63
	Forth	-0.34	***	0.38	-0.96	***	0.56
	Tay	-0.16	***	0.34	0.17	***	0.60
B2*Visitor	All	-0.31	***	0.23	1.63	***	0.24
	Clyde	-0.81	***	0.41	-1.38	***	0.50
	Forth	-0.09	***	0.42	0.32	***	0.44
	Tay	0.05	***	0.40	0.27	***	0.37
R1*Visitor	All	0.22	***	0.15	-0.01	***	0.52
	Clyde	-0.01	***	0.27	0.12	***	0.62
	Forth	0.48	***	0.27	0.06	***	0.49
	Tay	0.29	***	0.24	0.14	***	0.78
R2*Visitor	All	0.46	***	0.15	-1.16	***	0.26
	Clyde	0.46	***	0.29	0.49	***	0.60
	Forth	0.41	***	0.29	0.39	***	0.42
	Tay	0.65	***	0.24	-0.54	***	0.68
Cost*Visitor	All	0.00	***	0.00	-	-	-
	Clyde	0.01	***	0.00	-	-	-
	Forth	0.00	***	0.00	-	-	-
	Tay	0.00	***	0.00	-	-	-
F1	All	1.76	***	0.15	0.71	***	0.20
	Clyde	2.26	***	0.29	0.41	***	0.55
	Forth	1.63	***	0.25	0.71	***	0.33
	Tay	1.54	***	0.25	0.94	***	0.33
F2	All	2.27	***	0.18	-1.03	***	0.18
	Clyde	2.86	***	0.37	1.43	***	0.33
	Forth	1.94	***	0.29	-0.76	***	0.36
	Tay	2.20	***	0.30	0.85	***	0.35
B1	All	1.93	***	0.16	-0.08	***	0.47
	Clyde	2.36	***	0.32	0.63	***	0.38
	Forth	1.87	***	0.28	0.03	***	0.46
	Tay	1.66	***	0.27	-0.11	***	0.40
B2	All	1.98	***	0.18	-0.94	***	0.16
	Clyde	2.17	***	0.33	1.07	***	0.31
	Forth	1.98	***	0.30	0.73	***	0.29
	Tay	1.94	***	0.30	-1.08	***	0.28
R1	All	0.52	***	0.11	-0.01	***	0.22
	Clyde	0.69	***	0.22	-0.11	***	0.37
	Forth	0.52	***	0.19	0.00	***	0.27
	Tay	0.36	***	0.18	0.14	***	0.64
R2	All	0.39	***	0.12	0.65	***	0.18
	Clyde	0.46	***	0.23	-0.79	***	0.34
	Forth	0.39	***	0.20	0.61	***	0.31
	Tay	0.30	***	0.18	0.53	***	0.33
Cost	All	-0.02	***	0.00	-	-	-

	RPL interacted 6						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Clyde	-0.02	***	0.00	-	-	-
	Forth	-0.02	***	0.00	-	-	-
	Tay	-0.01	***	0.00	-	-	-
ASC	All	-1.85	***	0.29	3.33	***	0.27
	Clyde	-1.94	***	0.57	4.02	***	0.55
	Forth	-1.74	***	0.50	3.31	***	0.47
	Tay	-1.79	***	0.45	2.68	***	0.37
Log-likelihood	All	-2742.63					
	Clyde	-889.22					
	Forth	-915.77					
	Tay	-902.93					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.28					
	Forth	0.27					
	Tay	0.29					
AIC	All	5541.26					
	Clyde	1834.44					
	Forth	1887.53					
	Tay	1861.86					
BIC	All	5714.02					
	Clyde	1976.11					
	Forth	2029.63					
	Tay	2004.10					

Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

	RPL interacted 7						
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1*Visitor*Resident	All	-0.63	***	0.21	-1.30	***	0.31
	Clyde	-1.47	***	0.36	-0.14	***	0.49
	Forth	-0.09	***	0.41	1.46	***	0.62
	Tay	0.12	***	0.61	0.69	***	0.79
F2*Visitor*Resident	All	-1.05	***	0.26	-0.26	***	0.27
	Clyde	-1.75	***	0.45	-0.65	***	0.42
	Forth	-0.88	***	0.50	-0.74	***	0.60
	Tay	0.06	***	0.78	1.63	***	0.97
B1*Visitor*Resident	All	-0.28	***	0.24	0.69	***	0.43
	Clyde	-0.67	***	0.40	-0.49	***	0.81
	Forth	-0.29	***	0.45	-0.95	***	1.08
	Tay	0.18	***	0.60	0.17	***	0.84
B2*Visitor*Resident	All	-0.38	***	0.25	-0.30	***	0.29
	Clyde	-0.84	***	0.40	-1.17	***	0.65
	Forth	-0.40	***	0.49	-0.23	***	0.55
	Tay	1.12	***	0.76	0.97	***	0.64
R1*Visitor*Resident	All	0.19	***	0.17	0.34	***	0.44
	Clyde	-0.02	***	0.28	0.24	***	0.86
	Forth	0.32	***	0.33	0.49	***	0.56
	Tay	-0.14	***	0.47	0.81	***	1.01
R2*Visitor*Resident	All	0.57	***	0.18	-0.10	***	0.29
	Clyde	0.62	***	0.29	1.01	***	0.69
	Forth	0.49	***	0.36	-0.49	***	0.48
	Tay	0.85	***	0.46	-0.55	***	0.95
Cost*Visitor*Resident	All	0.01	***	0.00	-	-	-
	Clyde	0.01	***	0.00	-	-	-
	Forth	0.01	***	0.00	-	-	-
	Tay	-0.01	***	0.01	-	-	-
F1	All	1.86	***	0.12	0.57	***	0.17
	Clyde	2.41	***	0.28	0.72	***	0.34
	Forth	1.82	***	0.23	-0.75	***	0.28
	Tay	1.61	***	0.17	0.49	***	0.26
F2	All	2.43	***	0.15	-1.06	***	0.14
	Clyde	3.03	***	0.36	1.56	***	0.31
	Forth	2.25	***	0.28	-1.14	***	0.27
	Tay	2.29	***	0.22	0.72	***	0.22
B1	All	1.78	***	0.13	-0.35	***	0.24
	Clyde	2.19	***	0.29	0.50	***	0.37
	Forth	1.82	***	0.25	0.71	***	0.27
	Tay	1.52	***	0.19	0.00	***	0.32
B2	All	1.94	***	0.14	0.90	***	0.13
	Clyde	2.08	***	0.30	0.94	***	0.28
	Forth	2.06	***	0.27	-0.88	***	0.27
	Tay	1.84	***	0.22	-0.94	***	0.19
R1	All	0.59	***	0.09	-0.01	***	0.19
	Clyde	0.70	***	0.19	-0.06	***	0.39
	Forth	0.70	***	0.17	0.03	***	0.25
	Tay	0.48	***	0.13	0.00	***	0.72
R2	All	0.49	***	0.09	-0.56	***	0.15
	Clyde	0.47	***	0.20	-0.73	***	0.31
	Forth	0.46	***	0.17	-0.70	***	0.25
	Tay	0.52	***	0.13	-0.26	***	0.37
Cost	All	-0.02	***	0.00	-	-	-

		RPL interacted 7					
Attribute	Dataset	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
	Clyde	-0.02	***	0.00	-	-	-
	Forth	-0.02	***	0.00	-	-	-
	Tay	-0.01	***	0.00	-	-	-
ASC	All	-1.86	***	0.29	3.35	***	0.27
	Clyde	-1.93	***	0.56	4.00	***	0.55
	Forth	-1.87	***	0.53	-3.47	***	0.50
	Tay	-1.83	***	0.45	2.72	***	0.39
Log-likelihood	All	-2730.29					
	Clyde	-878.93					
	Forth	-913.23					
	Tay	-900.99					
Observations	All	3534.00					
	Clyde	1164.00					
	Forth	1182.00					
	Tay	1188.00					
Adjusted rho-sq	All	0.29					
	Clyde	0.29					
	Forth	0.28					
	Tay	0.29					
AIC	All	5516.59					
	Clyde	1813.87					
	Forth	1882.46					
	Tay	1857.99					
BIC	All	5689.35					
	Clyde	1955.54					
	Forth	2024.56					
	Tay	2000.23					

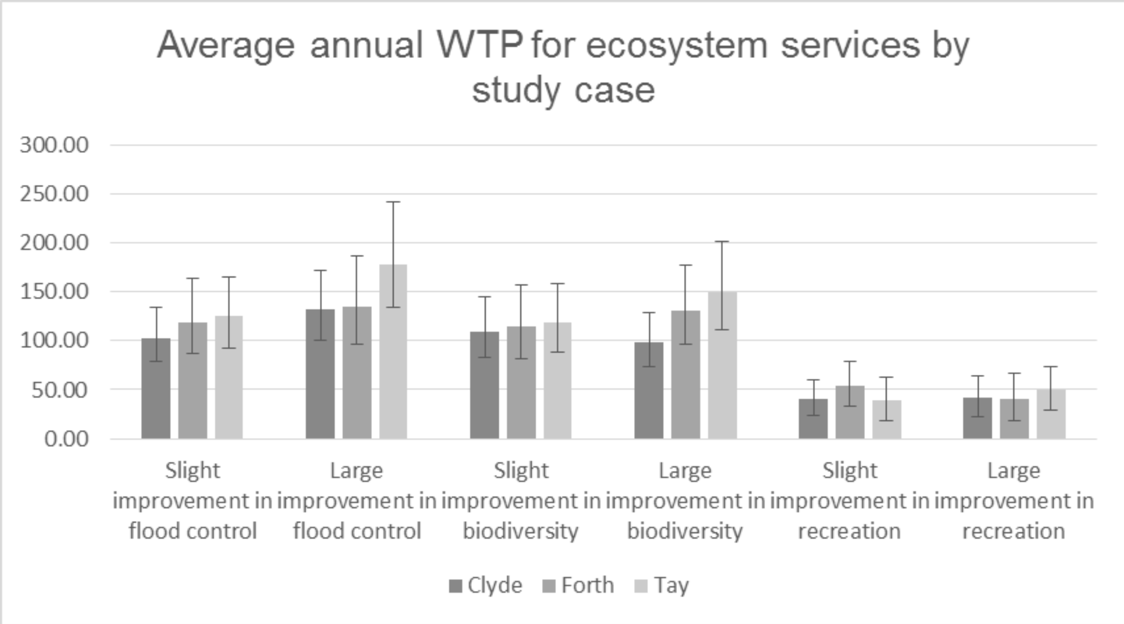
Source: Scottish estuarine management Choice Experiment, 2016.
Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels.
Standard errors computed by the Delta method.
Rows present parameter estimates for the pooled dataset, as well as each case study.

Annex 11 Welch two sample t-tests summary table

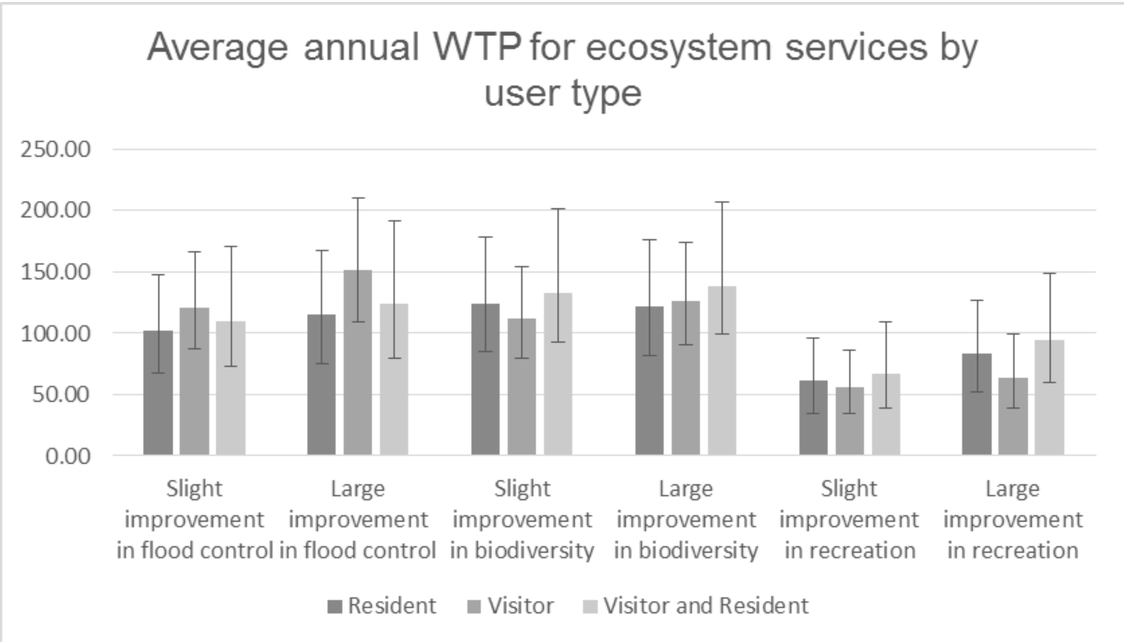
	Variables	Type of t-test	T statistic	P value
1	wtppooledf1 and wtppooledb1	greater	-2.41	0.992
2	wtppooledf1 and wtppooledr1	greater	96.49	< 2.22e-16
3	wtppooledf2 and wtppooledb2	greater	12.75	< 2.22e-16
4	wtppooledf2 and wtppooledr2	greater	66.62	< 2.22e-16
5	wtppooledb1 and wtppooledr1	greater	238.2	< 2.22e-16
6	wtppooledb2 and wtppooledr2	greater	71.72	< 2.22e-16
7	wtppooledb1resident and wtppooledf1resident	greater	18.7	< 2.22e-16
8	wtppooledb1resident and wtppooledr1resident	greater	93.02	< 2.22e-16
9	wtppooledb2resident and wtppooledf2resident	greater	3.31	0.00047587
10	wtppooledb2resident and wtppooledr2resident	greater	33.96	< 2.22e-16
11	wtppooledf1visitor and wtppooledb1visitor	greater	9.63	< 2.22e-16
12	wtppooledf1visitor and wtppooledr1visitor	greater	94.59	< 2.22e-16
13	wtppooledf2visitor and wtppooledb2visitor	greater	11.81	< 2.22e-16
14	wtppooledf2visitor and wtppooledr2visitor	greater	48.14	< 2.22e-16
15	wtptayf1 and wtpforthf1	greater	-8.81	1
16	wtpforthf1 and wtpclydef1	greater	-3.91	0.99995
17	wtptayf2 and wtpforthf2	greater	10.81	< 2.22e-16
18	wtpforthf2 and wtpclydef2	greater	-13.1	1
19	wtptayb1 and wtpforthb1	greater	-13.03	1
20	wtpforthb1 and wtpclydeb1	greater	-24.5	1
21	wtptayb2 and wtpforthb2	greater	-0.74	0.77131
22	wtpforthb2 and wtpclydeb2	greater	11.25	< 2.22e-16
23	wtptayr1 and wtpforthr1	greater	-154.99	1
24	wtpforthr1 and wtpclyder1	greater	72.92	< 2.22e-16

	Variables	Type of t-test	T statistic	P value
25	wtptayr2 and wtpforthr2	greater	5.04	2.4916e-07
26	wtpforthr2 and wtpclyder2	greater	-10.8	1
27	wtppooledf1resident and wtppooledf1nonresident	greater	-13.08	1
28	wtppooledf2resident and wtppooledf2nonresident	greater	-20.13	1
29	wtppooledb1resident and wtppooledb1nonresident	greater	17.16	< 2.22e-16
30	wtppooledb2resident and wtppooledb2nonresident	greater	-3.06	0.9989
31	wtppooledr1resident and wtppooledr1nonresident	greater	75.7	< 2.22e-16
32	wtppooledr2resident and wtppooledr2nonresident	greater	48.58	< 2.22e-16
33	wtppooledf1visitor and wtppooledf1nonvisitor	greater	11.7	< 2.22e-16
34	wtppooledf2visitor and wtppooledf2nonvisitor	greater	4.11	2.0365e-05
35	wtppooledb1visitor and wtppooledb1nonvisitor	greater	-15.42	1
36	wtppooledb2visitor and wtppooledb2nonvisitor	greater	4.01	3.0468e-05
37	wtppooledr1visitor and wtppooledr1nonvisitor	greater	388.36	< 2.22e-16
38	wtppooledr2visitor and wtppooledr2nonvisitor	greater	45.84	< 2.22e-16

Annex 12 Site-specific WTP estimates and 95% CI for ES improvements (GBP/year)



Annex 13 User-specific WTP estimates and 95% CI for ES improvements (GBP/year)



Annex 14 Summary statistics of respondents and their households (n=571)

Variable	Mean	S.D.
Income (net, in £ per month)	1820.76	1127.51
Age	50.23	16.25
Household size	2.34	1.26
Gender (% female)	53.77	
Education (% with university degree and above)	40.46	
Employment (% economically active)	61.12	
Residency in the area (% residents)	31.70	
Visited the area for outdoor recreational activities (% visitors)	53.24	
People perceiving a better environmental status in the area than 10 years ago (% respondents)	18.56	
People perceiving a worse environmental status in the area than 10 years ago (% respondents)	19.61	

Annex 15 Sensitivity analysis for global spatial autocorrelation of WTP estimates for ES improvements

	P values of global Moran's I statistic													
	k=2		k=8		k=15		k=23		k=30		k=50		k=100	
Flood control														
Slight improvement	0.08	+	0.02	*	0.01	*	0.00	**	0.01	*	0.23		0.90	
Large improvement	0.78		0.80		0.67		0.54		0.61		0.15		0.26	
Biodiversity														
Slight improvement	0.67		0.23		0.40		0.40		0.54		0.39		0.57	
Large improvement	0.68		0.78		0.78		0.81		0.86		0.70		0.54	
Recreation														
Slight improvement	0.32		0.89		0.80		0.86		0.57		0.95		0.84	
Large improvement	0.05	*	0.49		0.64		0.80		0.82		0.37		0.13	

Annex 16 Proportion of significant clusters of WTP estimates for ES improvements

	Proportion of Significant Local Clusters						
	k=2	k=8	k=15	k=23	k=30	k=50	k=100
Flood control							
Slight improvement	0.06	0.09	0.10	0.09	0.09	0.06	0.01
Large improvement	0.05	0.05	0.04	0.05	0.04	0.07	0.06
Biodiversity							
Slight improvement	0.03	0.04	0.04	0.03	0.04	0.04	0.03
Large improvement	0.04	0.02	0.04	0.03	0.04	0.04	0.03
Recreation							
Slight improvement	0.03	0.03	0.03	0.02	0.03	0.01	0.01
Large improvement	0.06	0.04	0.03	0.05	0.04	0.06	0.07

Each row represents the total number significant clusters divided by the total sample.

Annex 17 Global spatial autocorrelation of resident’s WTP estimates for ES improvements

	All (n=181)		Tay (n=32)		Clyde (n=84)		Forth (n=70)	
Attribute	Moran's I	P values	Moran's I	P values	Moran's I	P values	Moran's I	P values
Flood control								
Slight improvement	0.03	(0.10) +	0.04	(0.18)	-0.03	(0.69)	-0.07	(0.89)
Large improvement	0.02	(0.14)	-0.12	(0.81)	-0.02	(0.57)	0.03	(0.16)
Biodiversity								
Slight improvement	0.00	(0.36)	-0.07	(0.87)	-0.03	(0.69)	-0.03	(0.65)
Large improvement	0.00	(0.46)	-0.13	(0.84)	0.01	(0.27)	-0.05	(0.78)
Recreation								
Slight improvement	-0.02	(0.75)	-0.04	(0.49)	0.01	(0.27)	-0.06	(0.82)
Large improvement	0.00	(0.39)	0.03	(0.21)	0.01	(0.34)	-0.01	(0.48)

Annex 18 Local spatial autocorrelation of resident’s WTP estimates for ES improvements

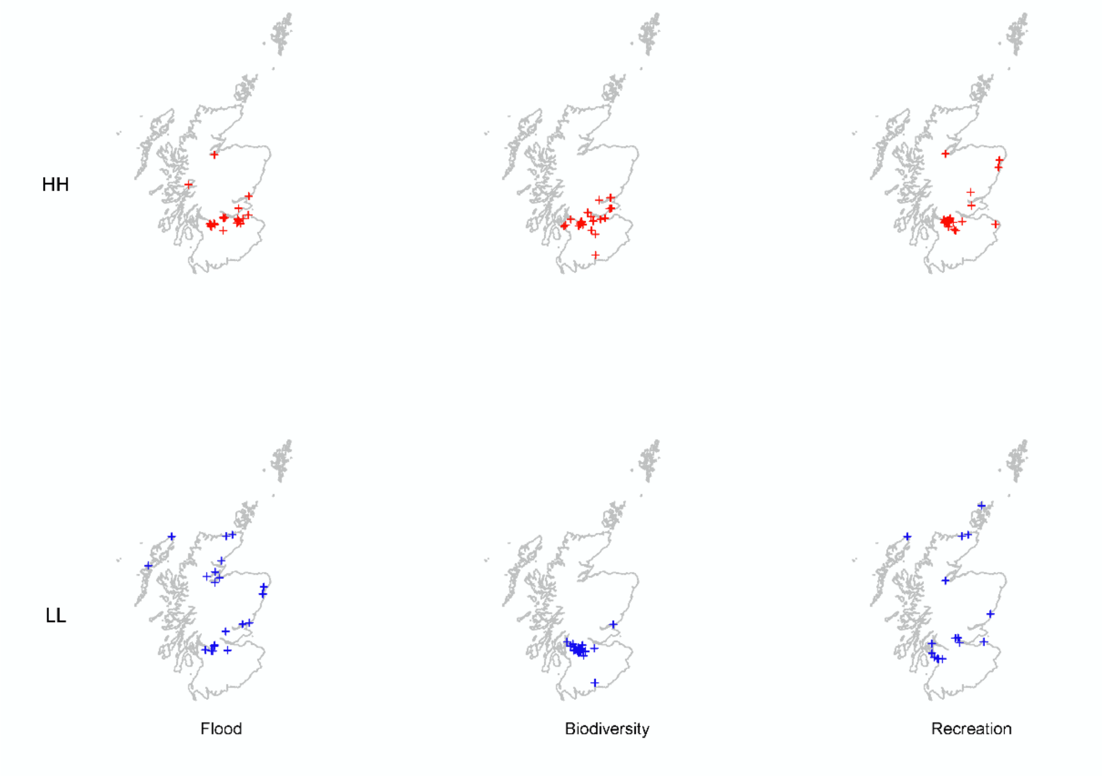
	HH				LL				HL				LH				NS			
Attribute	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay
Flood control																				
Slight improvement	9	1	0	2	3	1	1	0	0	0	0	0	0	0	0	0	169	82	69	25
Large improvement	7	5	5	0	8	0	6	0	0	0	0	0	0	0	0	0	166	79	59	27
Biodiversity																				
Slight improvement	1	2	0	0	6	2	2	0	0	0	0	0	0	0	0	0	174	80	68	27
Large improvement	6	4	0	0	2	0	0	0	0	0	0	0	0	0	0	0	173	80	70	27
Recreation																				
Slight improvement	1	3	1	0	5	1	0	0	0	0	0	0	0	0	0	0	175	80	69	27
Large improvement	5	4	2	1	2	1	1	1	0	0	0	0	0	0	0	0	174	79	67	25

Annex 19 Percentage of local clusters of WTP estimates for ES improvements

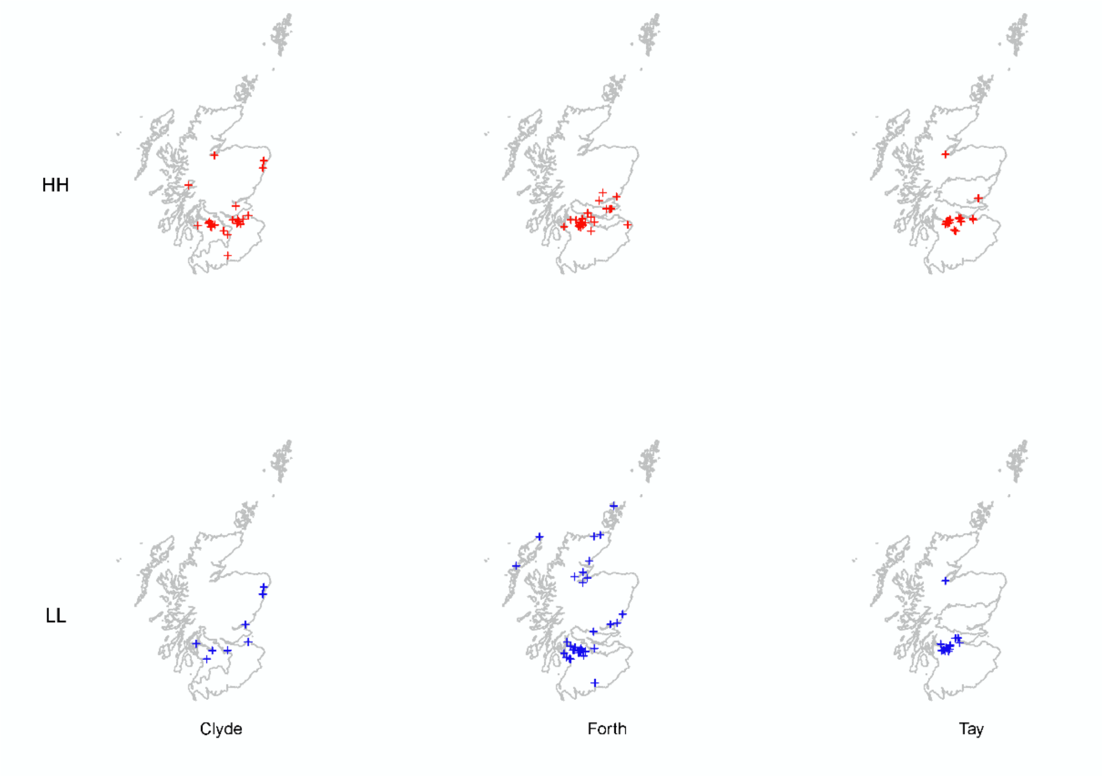
	HH				LL				HL				LH				NS			
Variables	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay	All	Clyde	Forth	Tay
Flood control																				
Slight improvement	4	3	1	2	5	1	3	2	0	0	0	0	0	0	0	0	91	97	96	96
Large improvement	2	5	2	1	3	2	6	0	0	0	0	0	0	0	0	0	95	94	93	99
Biodiversity																				
Slight improvement	1	1	2	1	2	1	6	0	0	0	0	0	0	0	0	0	97	98	92	99
Large improvement	2	2	6	2	1	1	3	5	0	0	0	0	0	0	0	0	97	97	91	93
Recreation																				
Slight improvement	1	3	1	3	1	1	1	1	0	0	0	0	0	0	0	0	98	96	98	96
Large improvement	2	0	4	3	2	1	4	1	0	0	0	0	0	0	0	0	95	99	92	96

HH=high-high, LL=low-low, HL=low-high, LH=high-low, NS=not significant; using k= 23 for neighbourhood definition.
This table contains the same information as table 5-5, but scale the numbers by dividing them by the total sample n=571
The sum of the percentages associated with the same dataset (e.g. ‘All’) per row leads to the total percentage (100%).

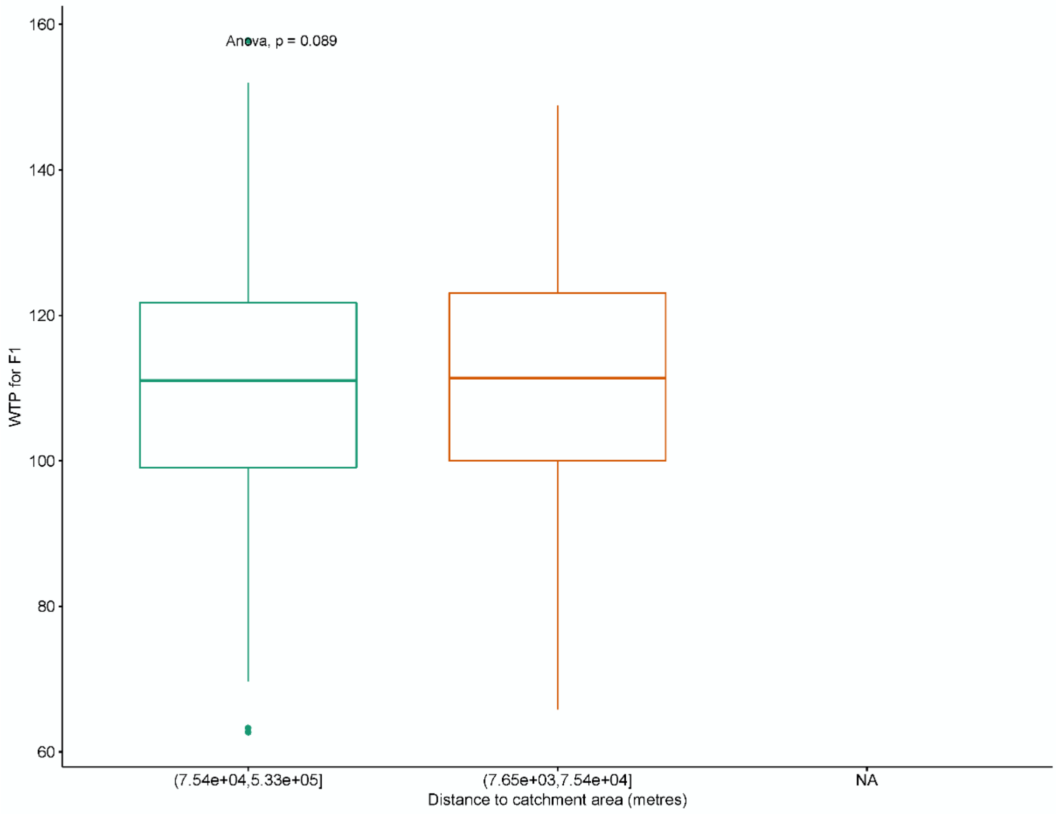
Annex 20 Local significant clusters of WTP estimates marked by ES

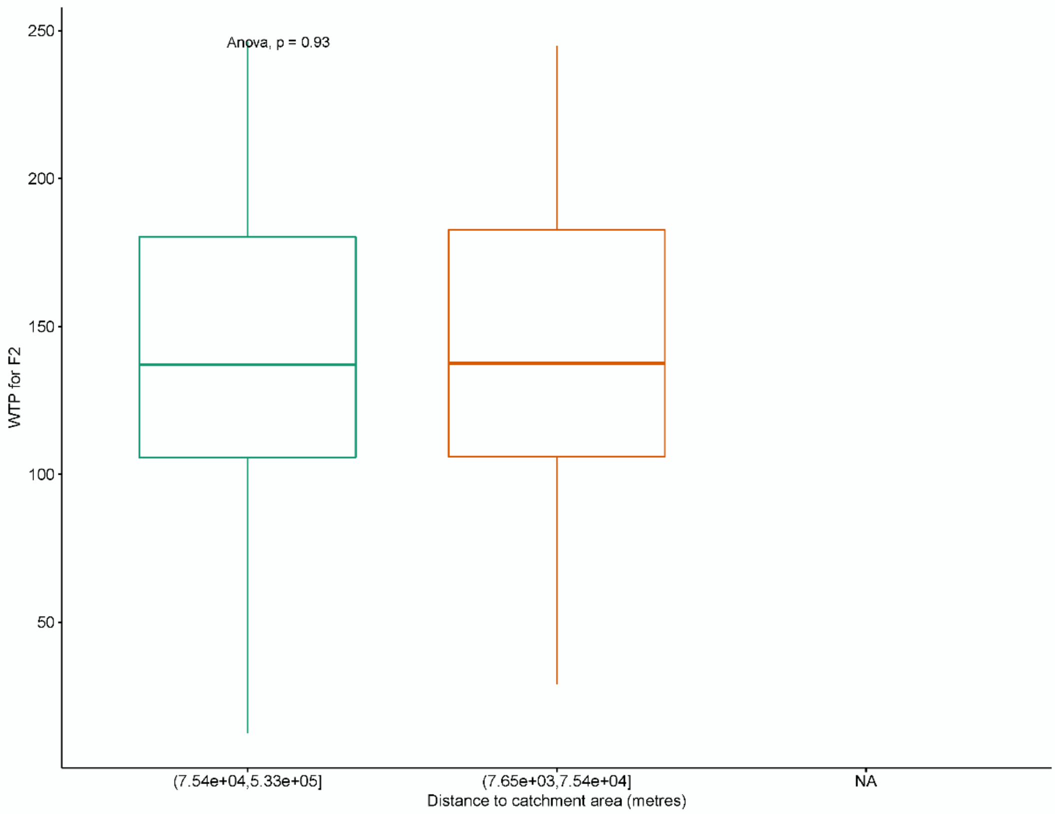


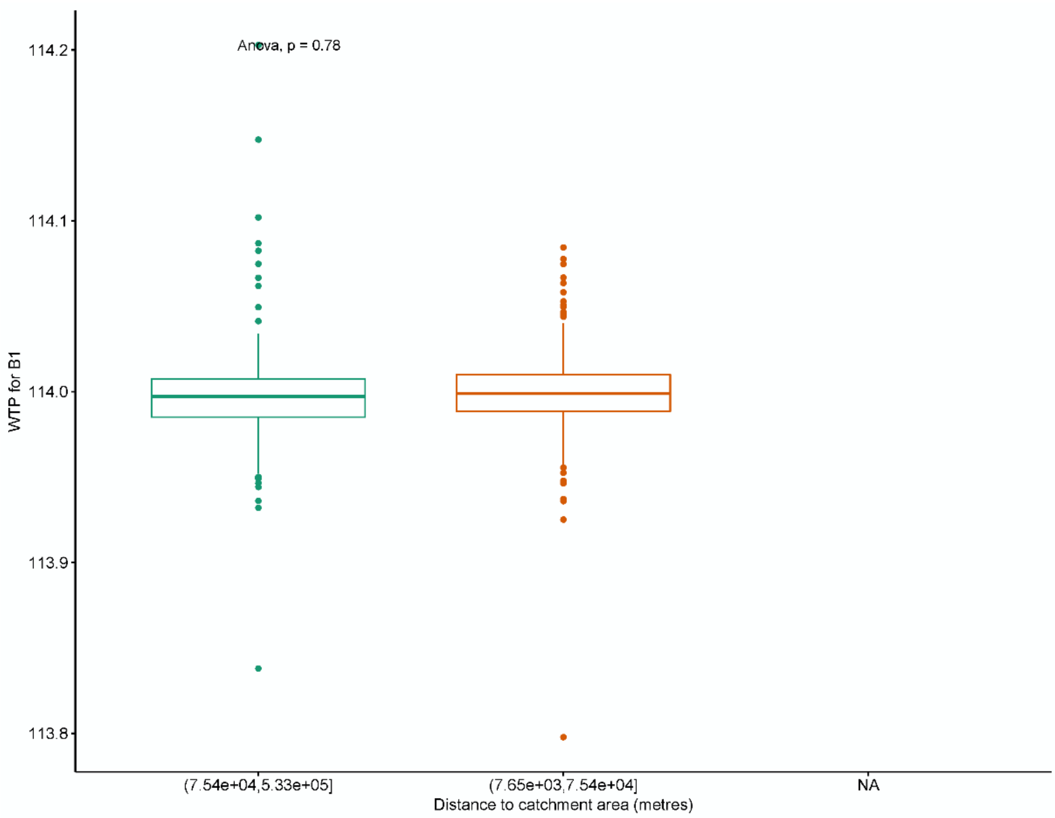
Annex 21 Local significant clusters of WTP estimates marked by survey

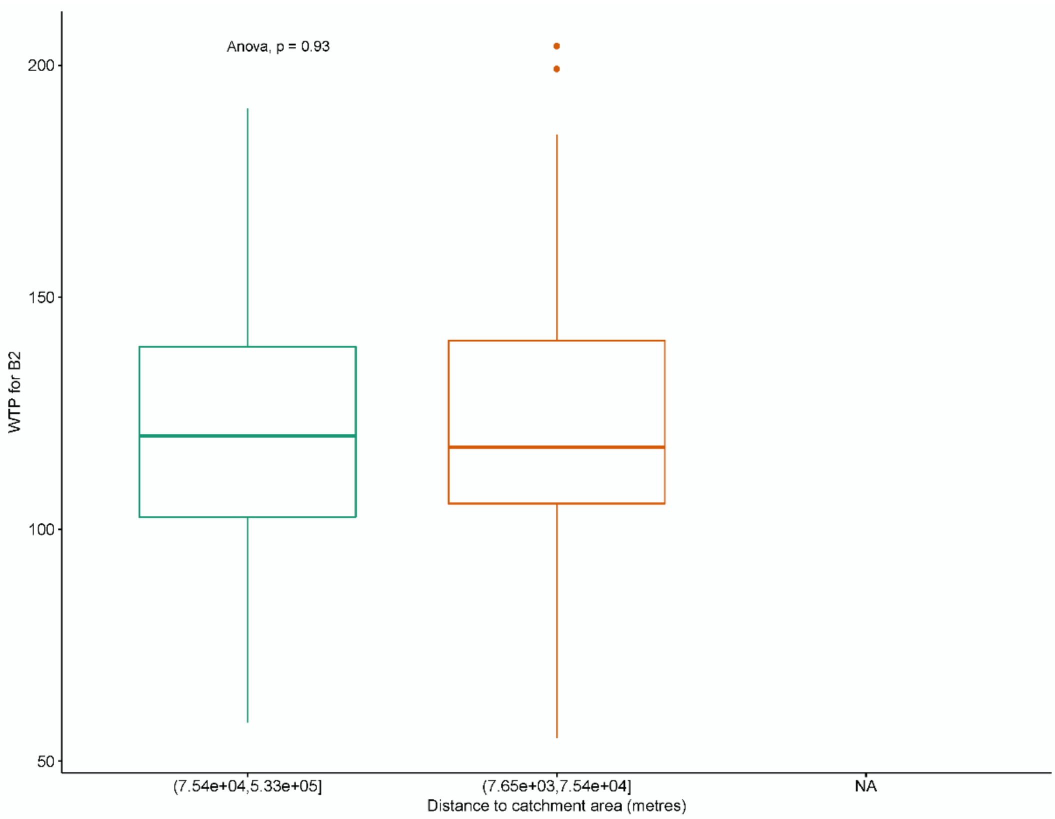


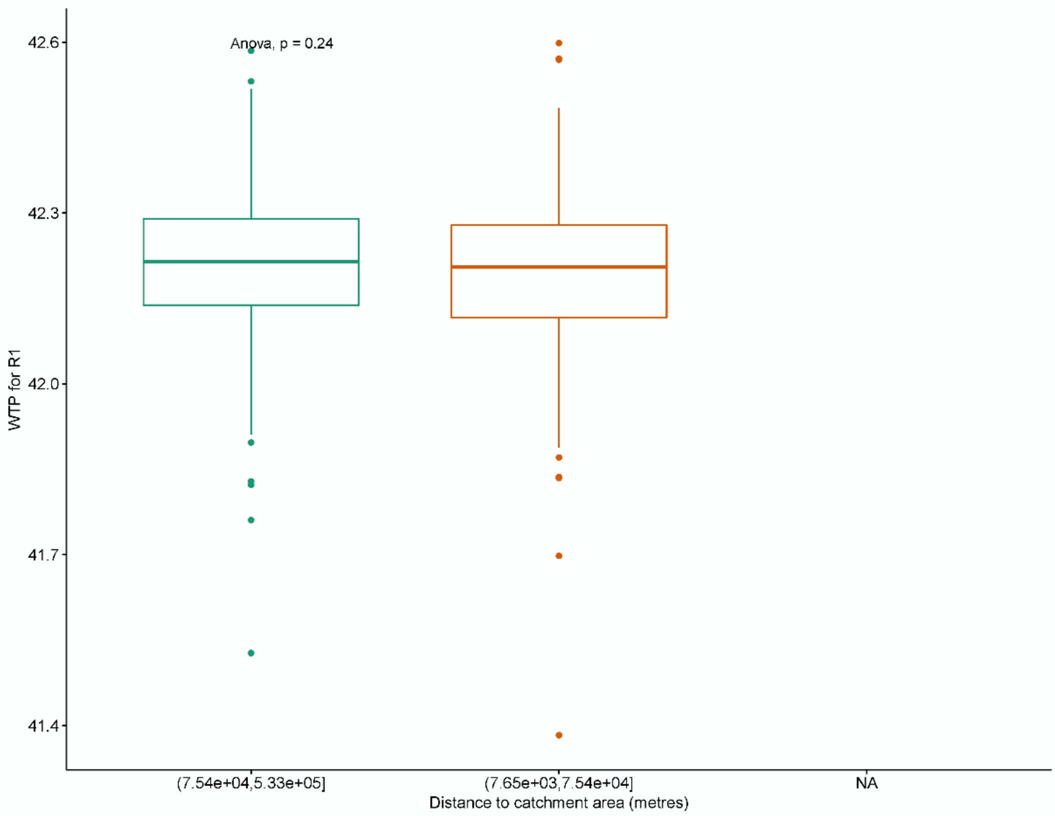
Annex 22 One-way ANOVA tests



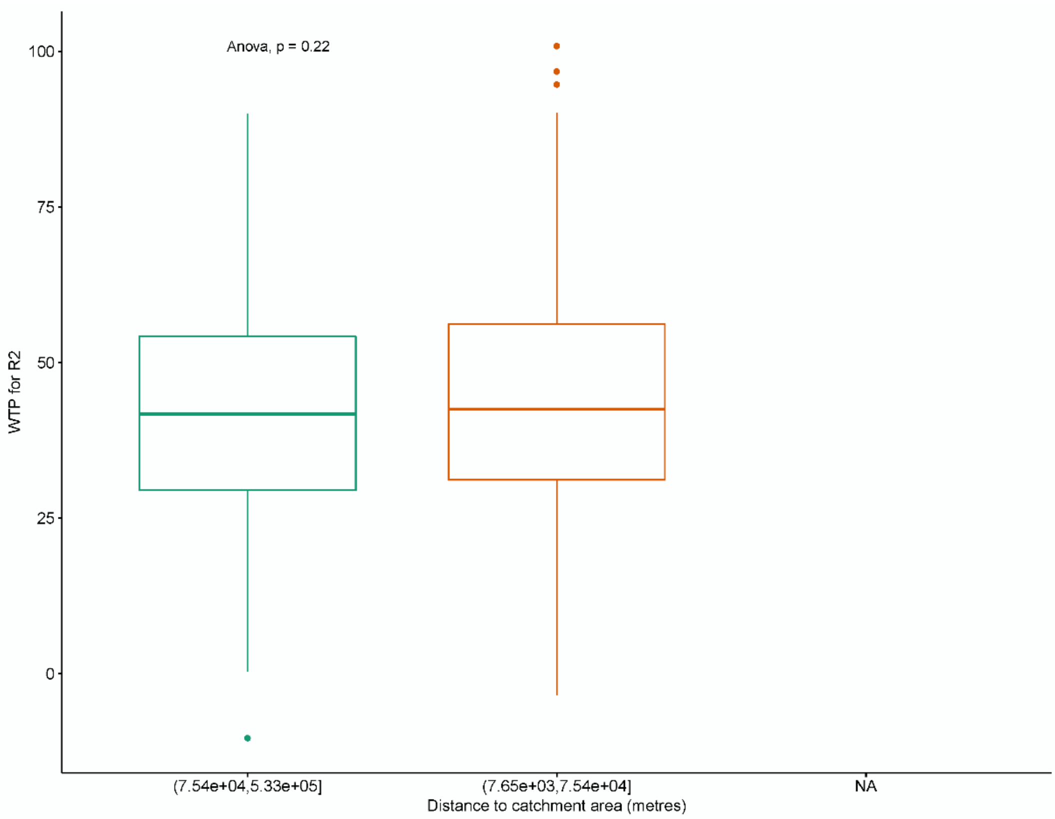



Anova, p = 0.78
114.2
114.1
114.0
113.9
113.8
WTP for B1
(7.54e+04,5.33e+05]
(7.65e+03,7.54e+04]
NA
Distance to catchment area (metres)


Anova, p = 0.93
200
150
100
50
WTP for B2
(7.54e+04,5.33e+05]
(7.65e+03,7.54e+04]
NA
Distance to catchment area (metres)

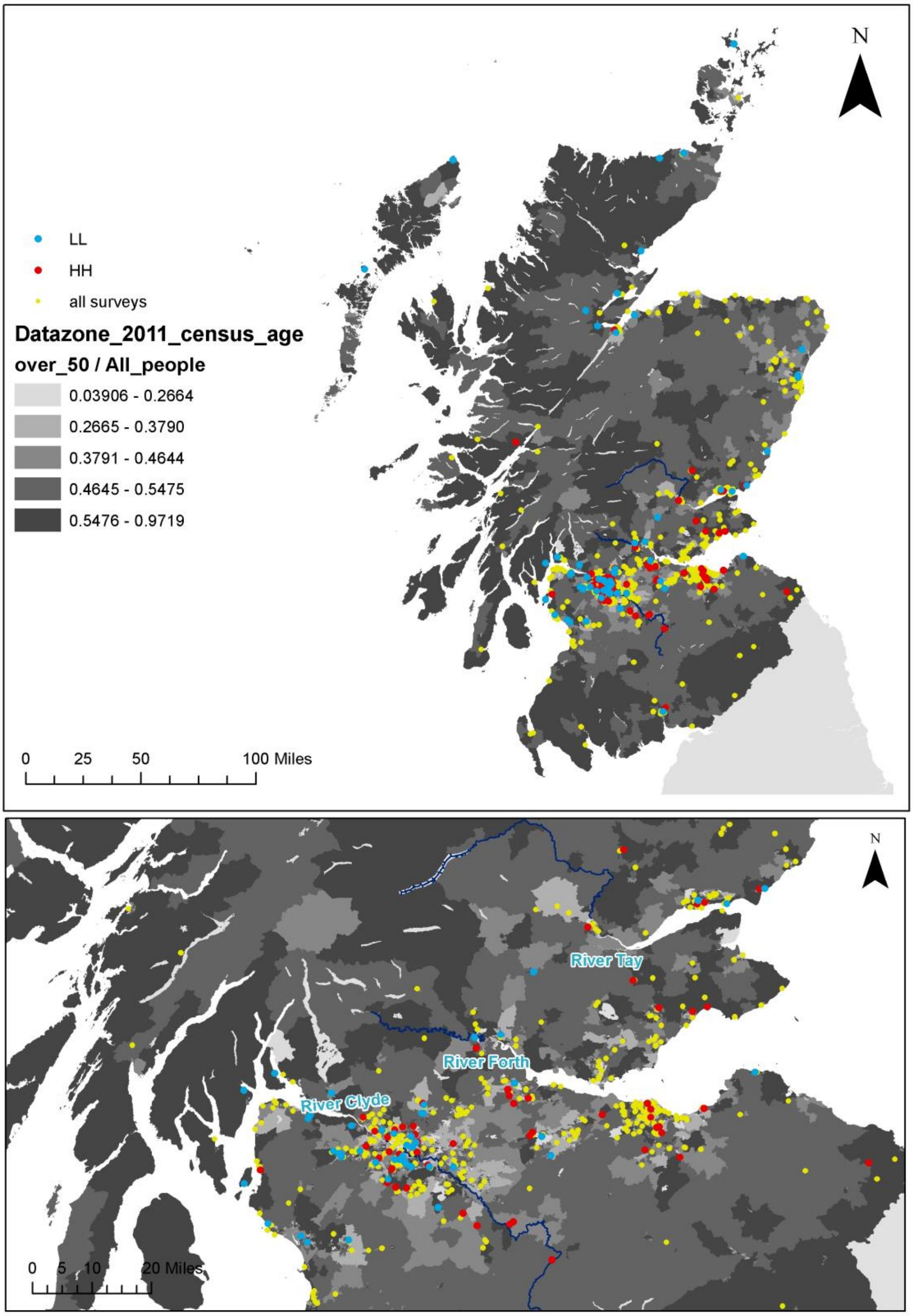


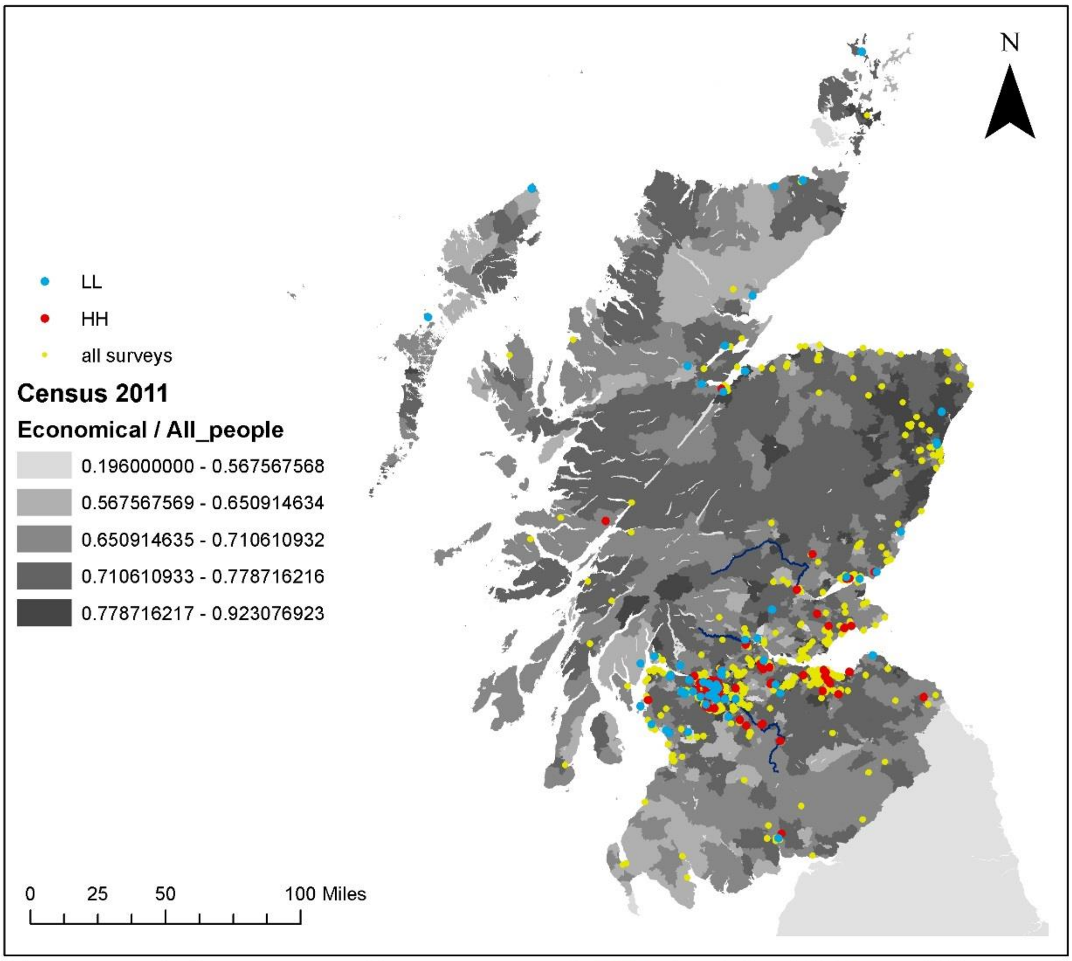
Anova, p = 0.24
42.6
42.3
42.0
41.7
41.4
WTP for R1
(7.54e+04,5.33e+05]
(7.65e+03,7.54e+04]
NA
Distance to catchment area (metres)



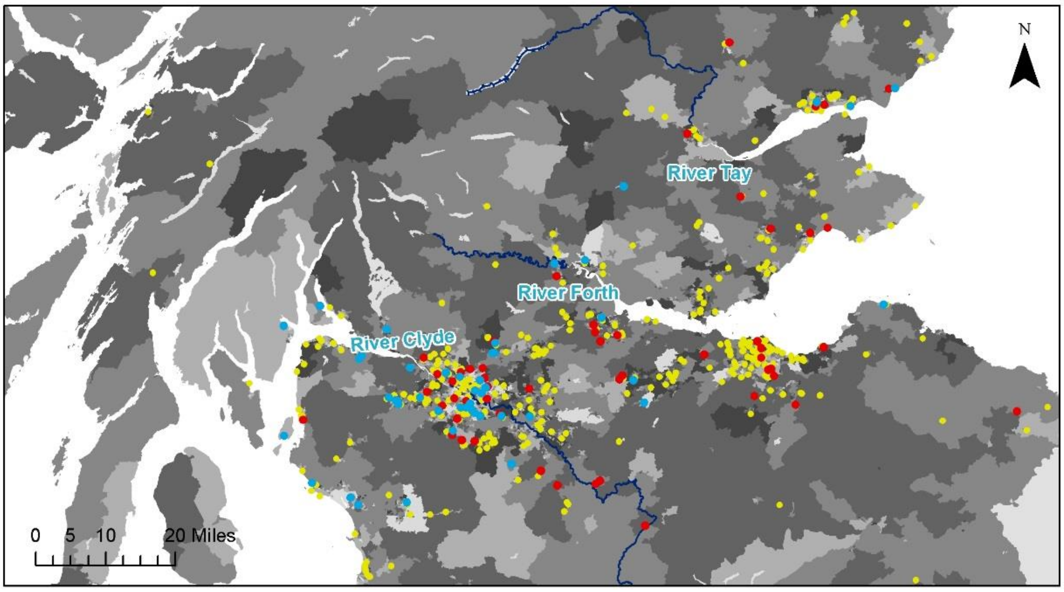
Anova, p = 0.22
100
75
50
25
0
WTP for R2
(7.54e+04,5.33e+05]
(7.65e+03,7.54e+04]
NA
Distance to catchment area (metres)

Annex 23 Local significant clusters of WTP estimates and socioeconomic indicators

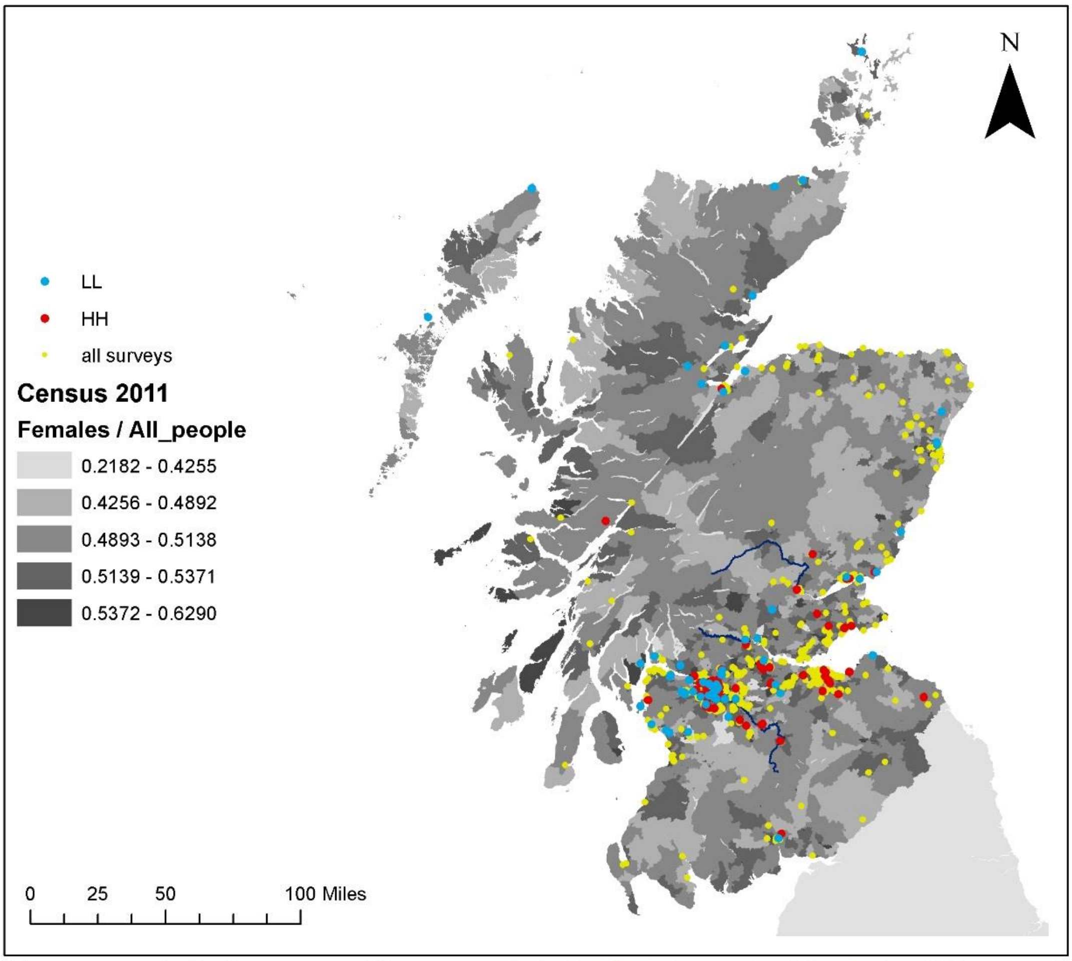


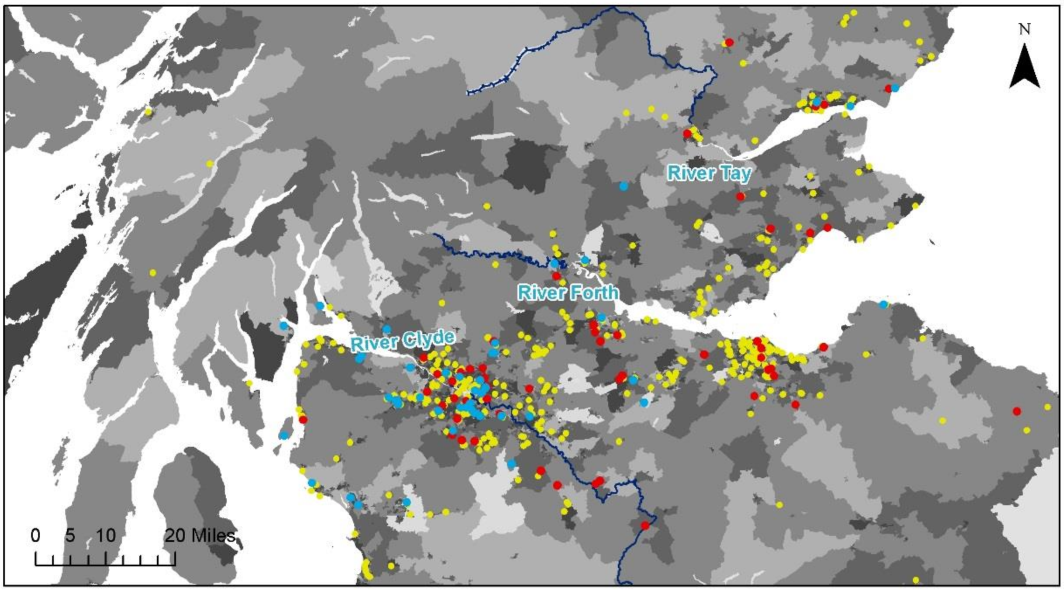


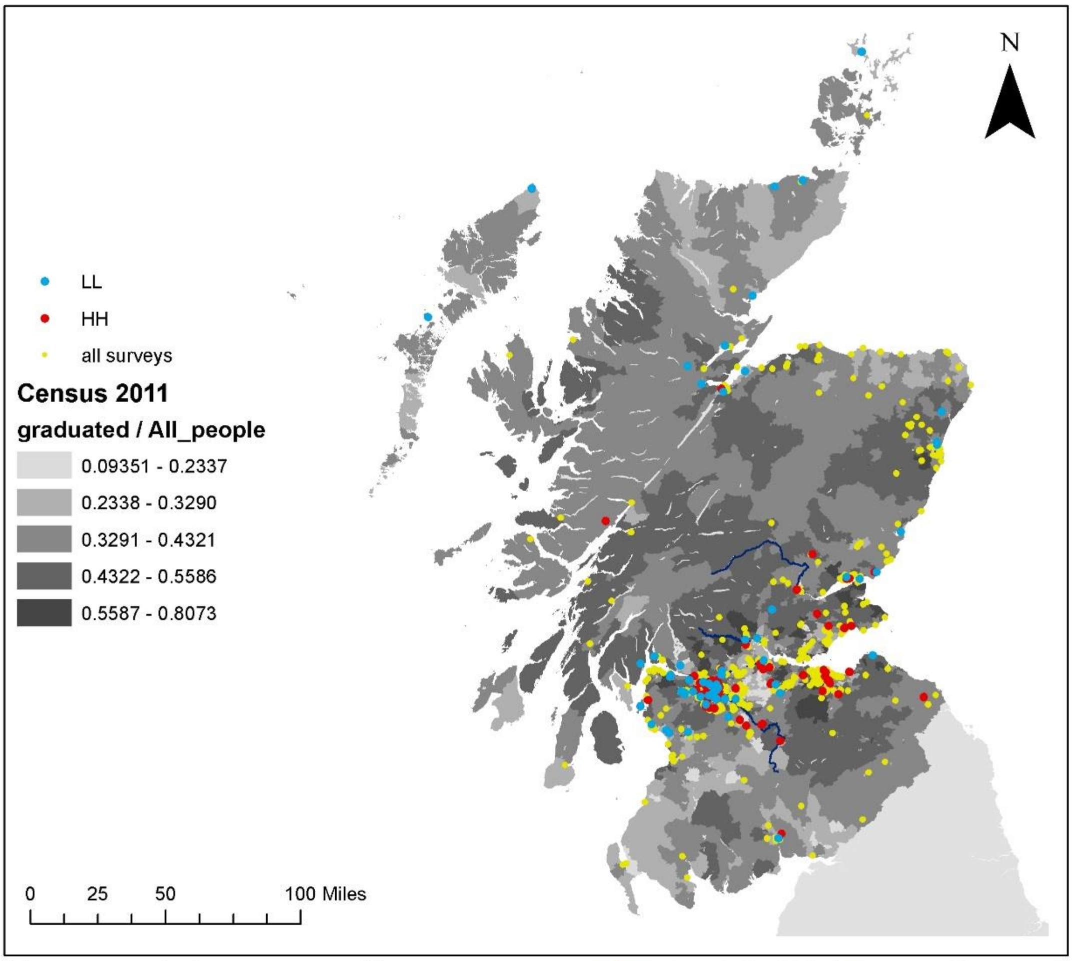
N
LL
HH
all surveys
Census 2011
Economical / All_people
0.196000000 - 0.567567568
0.567567569 - 0.650914634
0.650914635 - 0.710610932
0.710610933 - 0.778716216
0.778716217 - 0.923076923
0 25 50 100 Miles

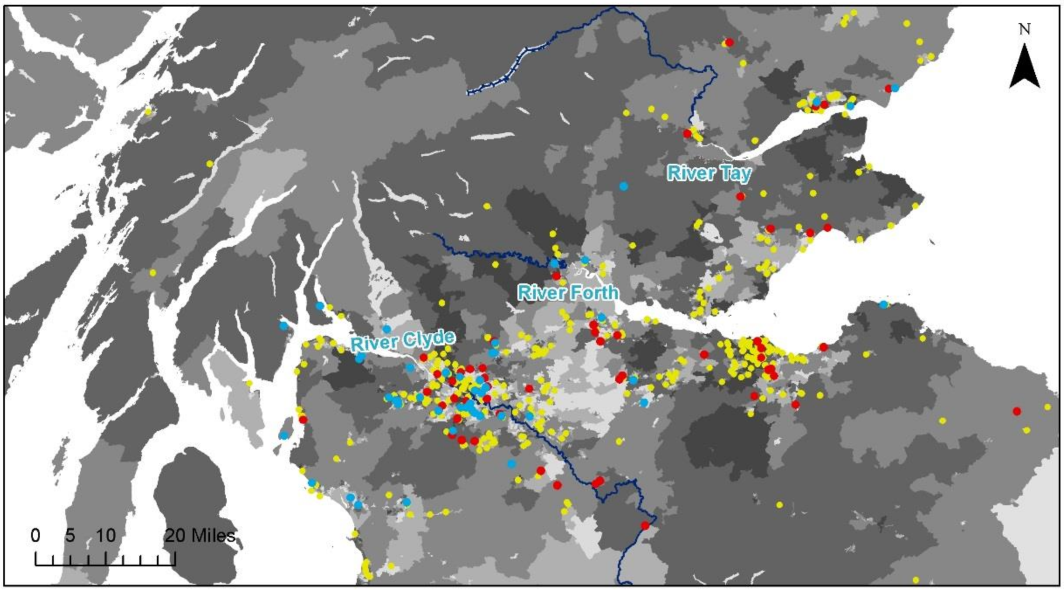


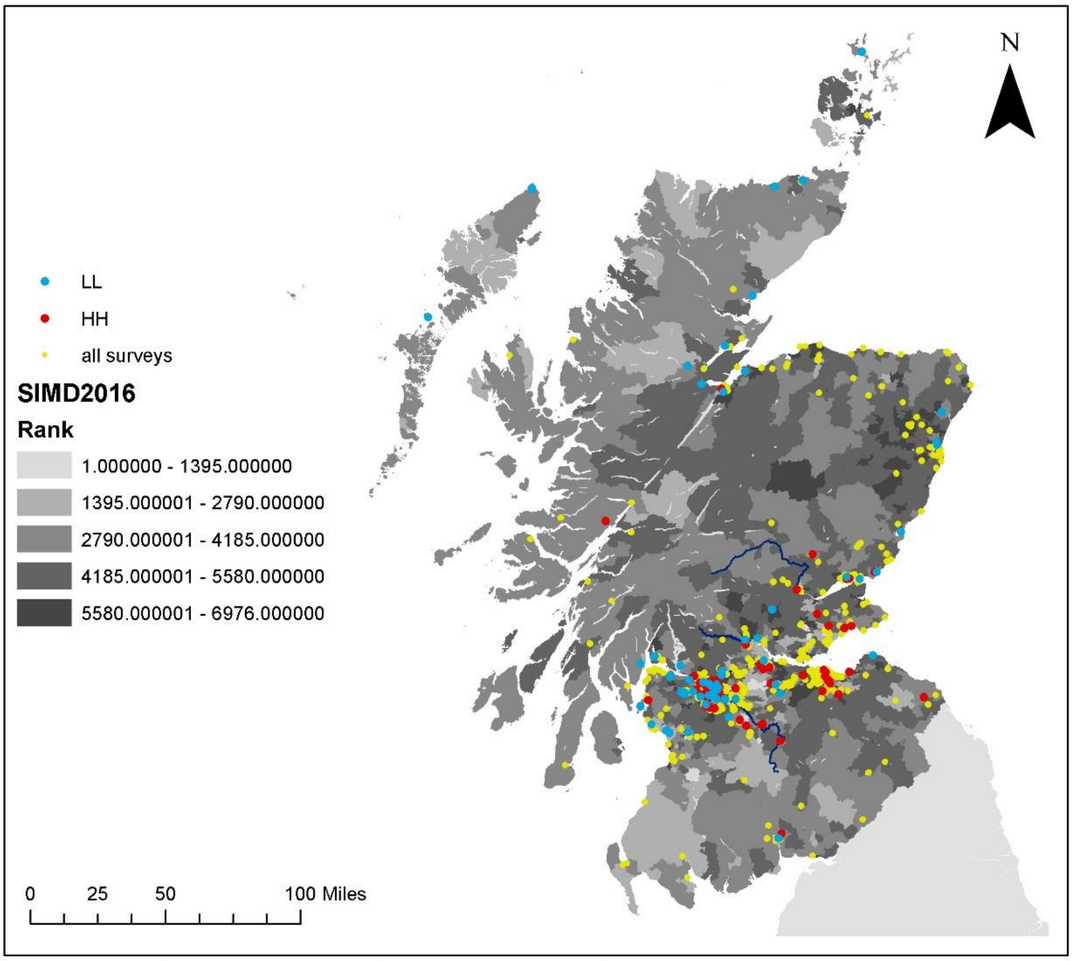
N
River Tay
River Forth
River Clyde
0 5 10 20 Miles

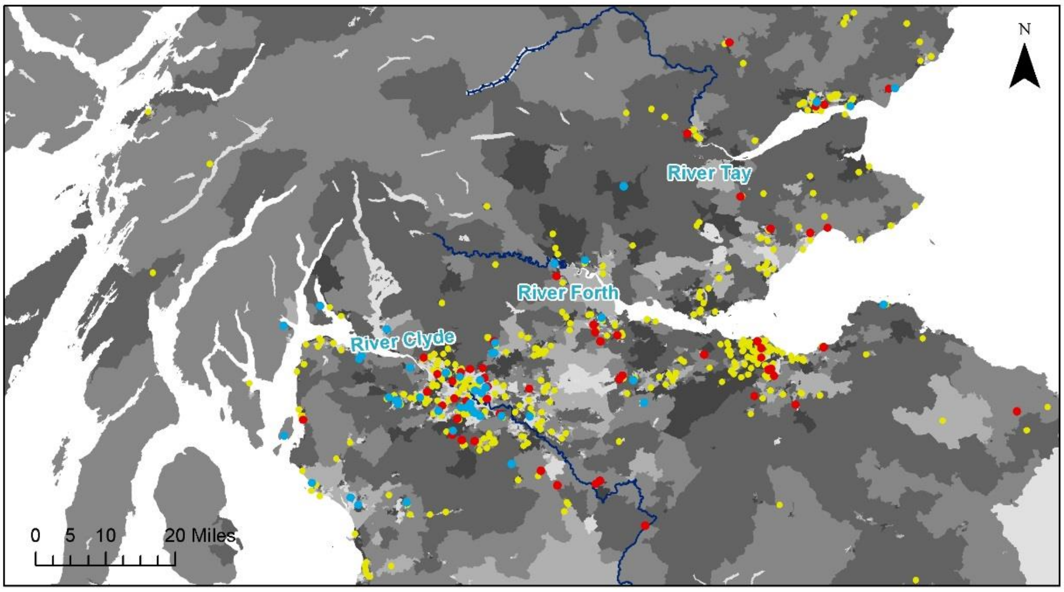

N
LL
HH
all surveys
Census 2011
Females / All_people
0.2182 - 0.4255
0.4256 - 0.4892
0.4893 - 0.5138
0.5139 - 0.5371
0.5372 - 0.6290
0 25 50 100 Miles


N
River Tay
River Forth
River Clyde
0 5 10 20 Miles

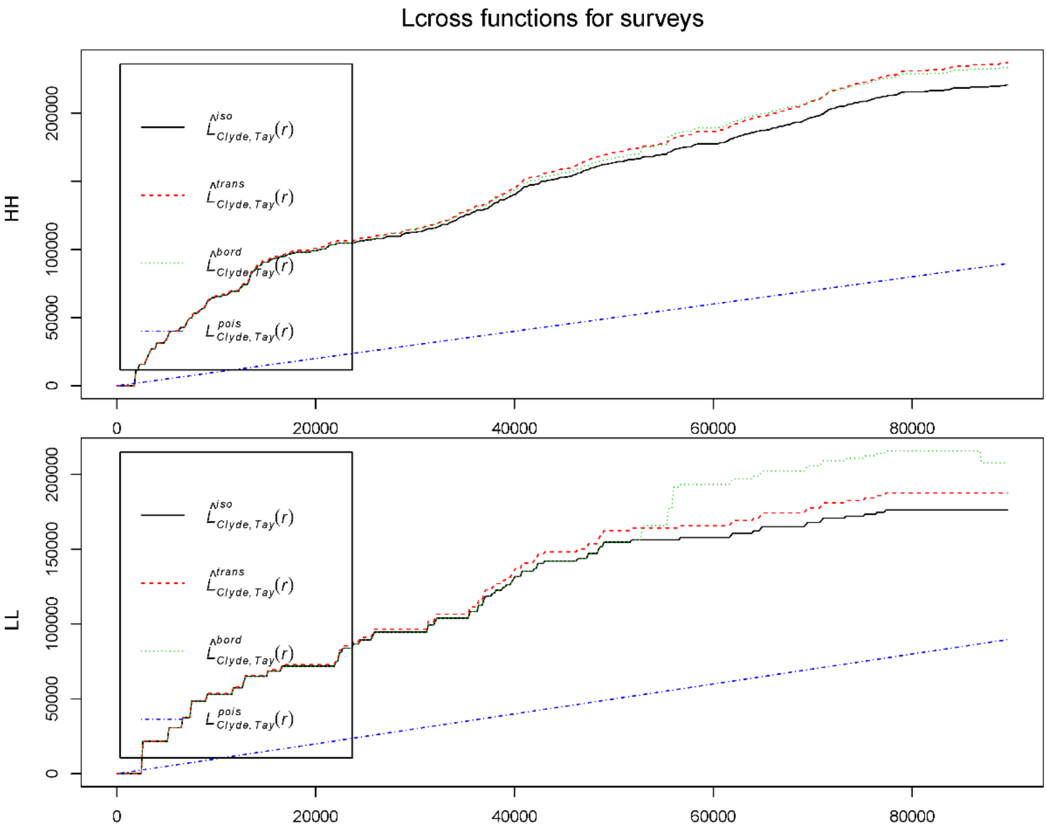

N
LL
HH
all surveys
Census 2011
graduated / All_people
0.09351 - 0.2337
0.2338 - 0.3290
0.3291 - 0.4321
0.4322 - 0.5586
0.5587 - 0.8073
0 25 50 100 Miles

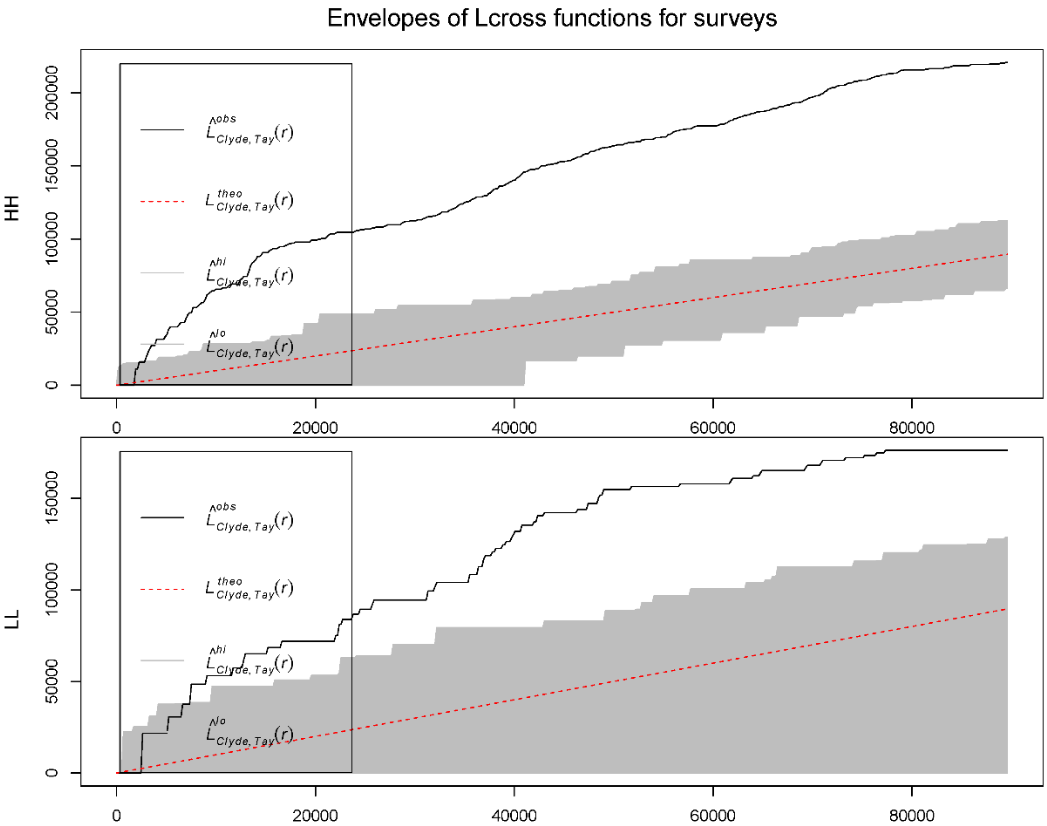

N
River Tay
River Forth
River Clyde
0 5 10 20 Miles

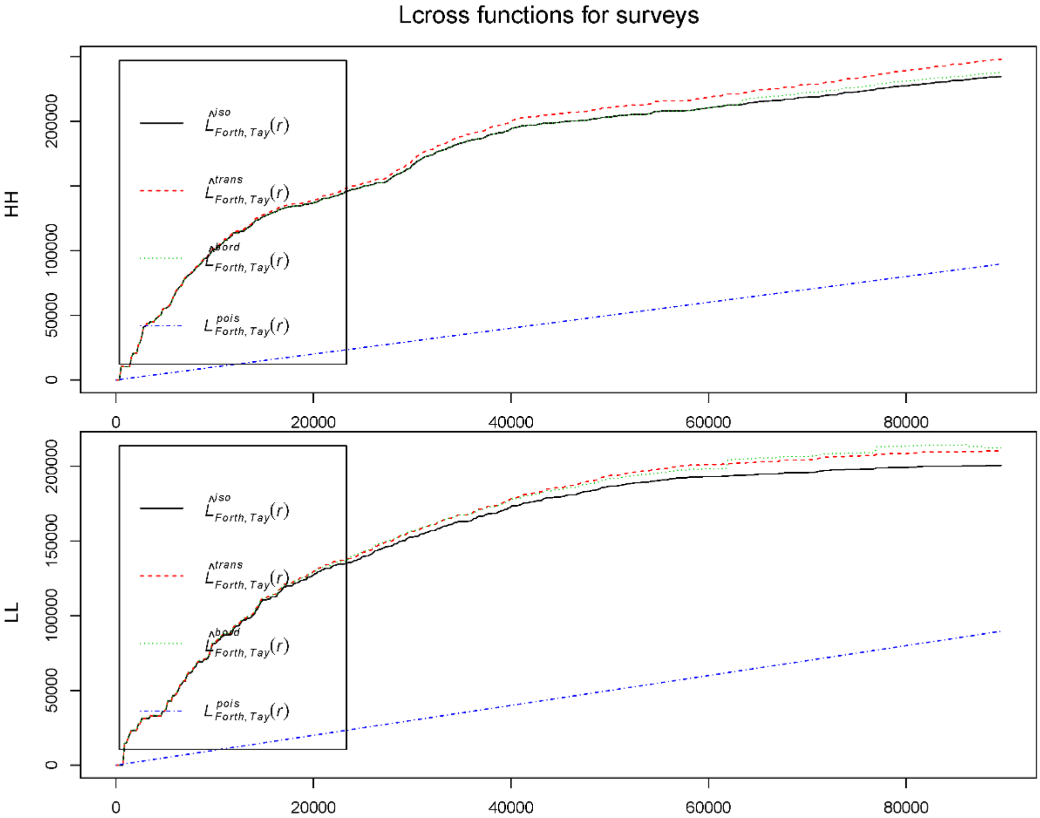

N
LL
HH
all surveys
SIMD2016
Rank
1.000000 - 1395.000000
1395.000001 - 2790.000000
2790.000001 - 4185.000000
4185.000001 - 5580.000000
5580.000001 - 6976.000000
0 25 50 100 Miles

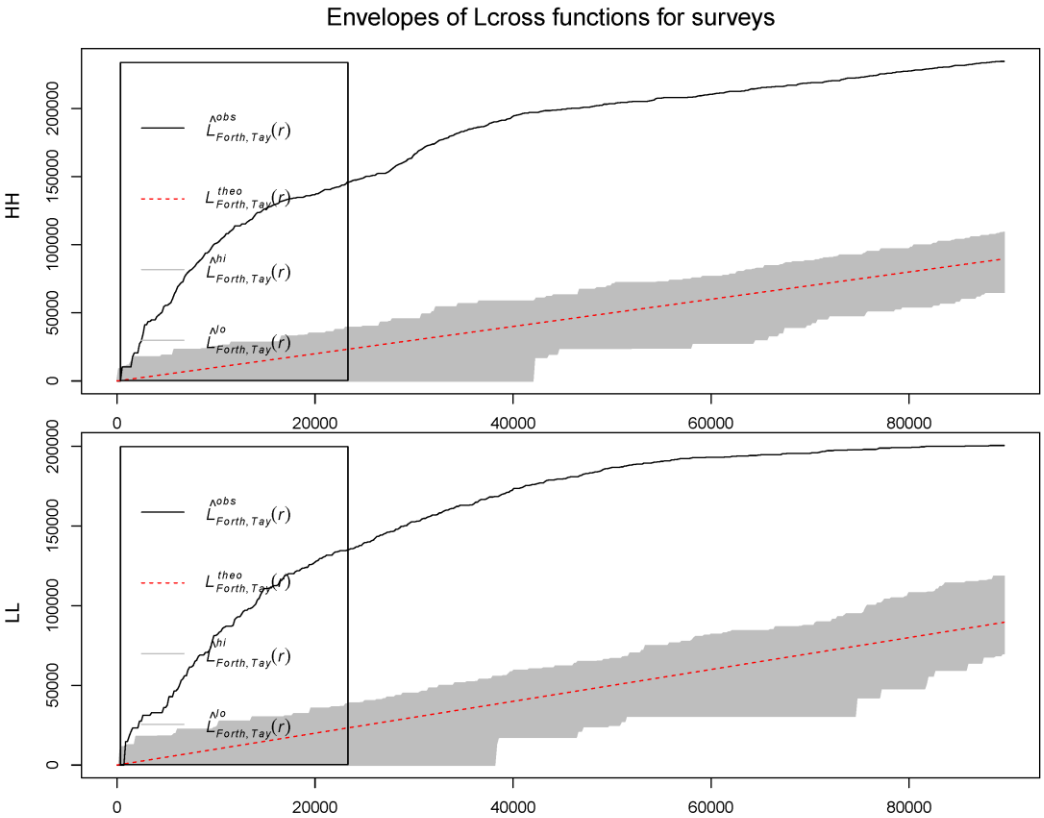

N
River Tay
River Forth
River Clyde
0 5 10 20 Miles

Annex 24 L-cross functions and envelopes for local clusters of WTP estimates marked by survey

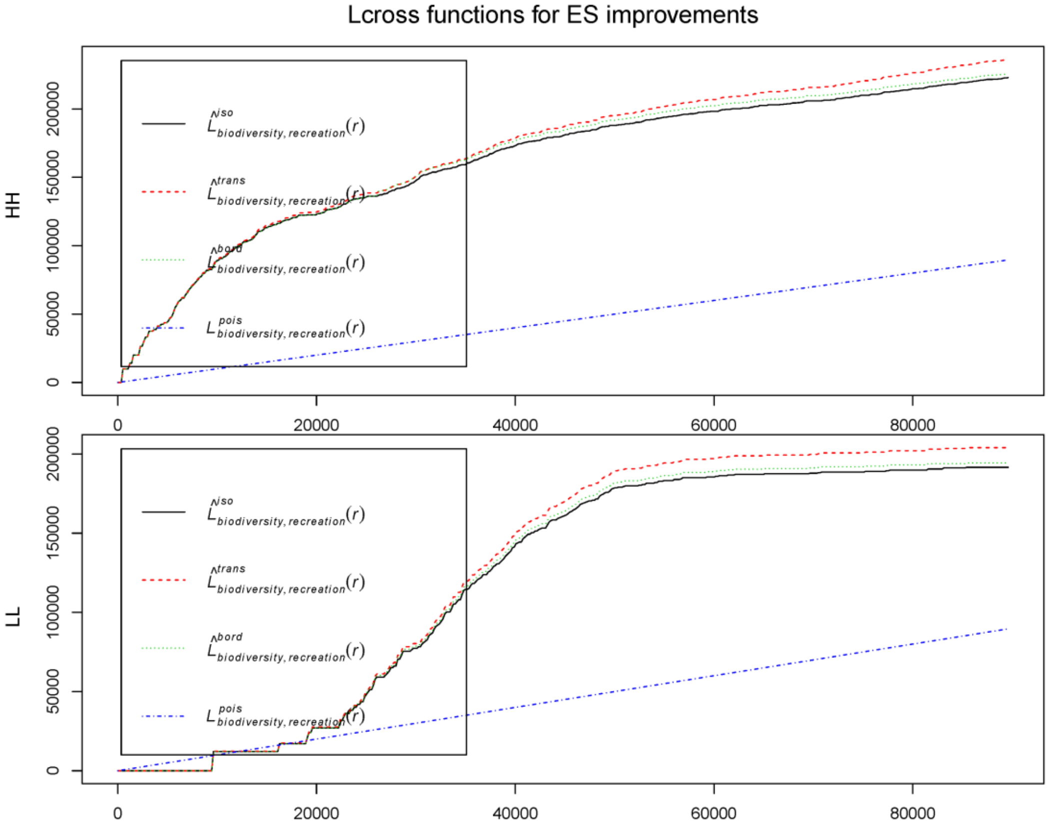


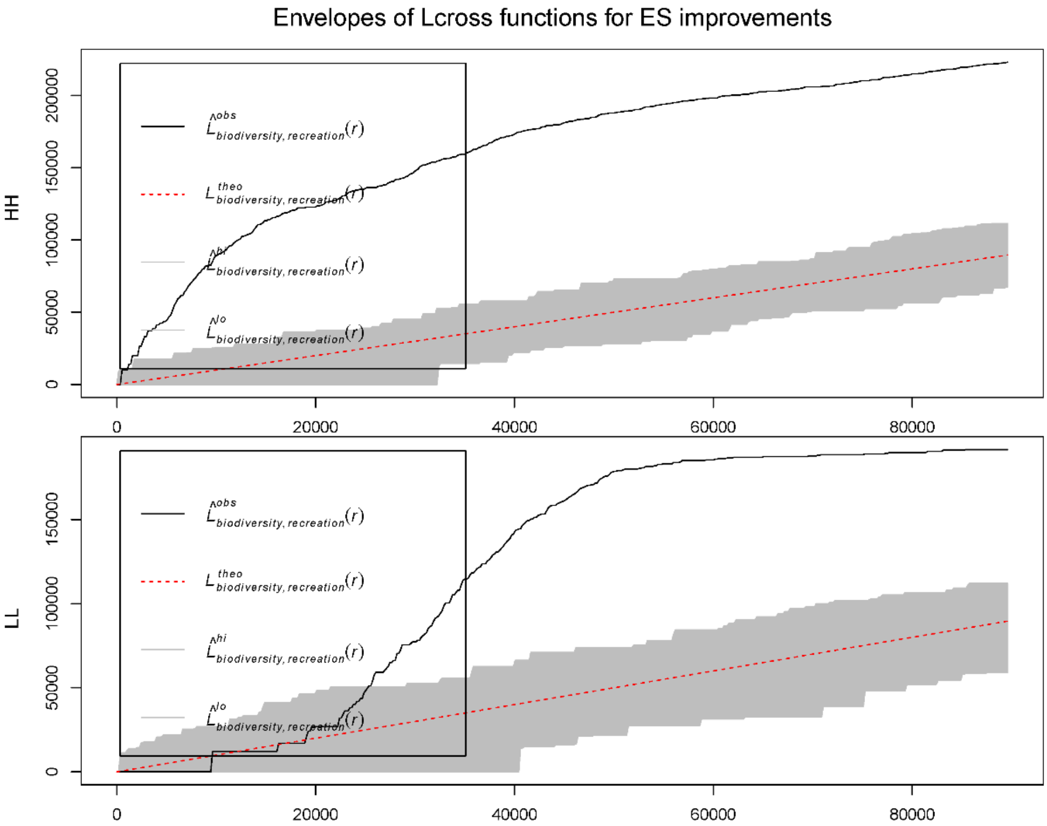


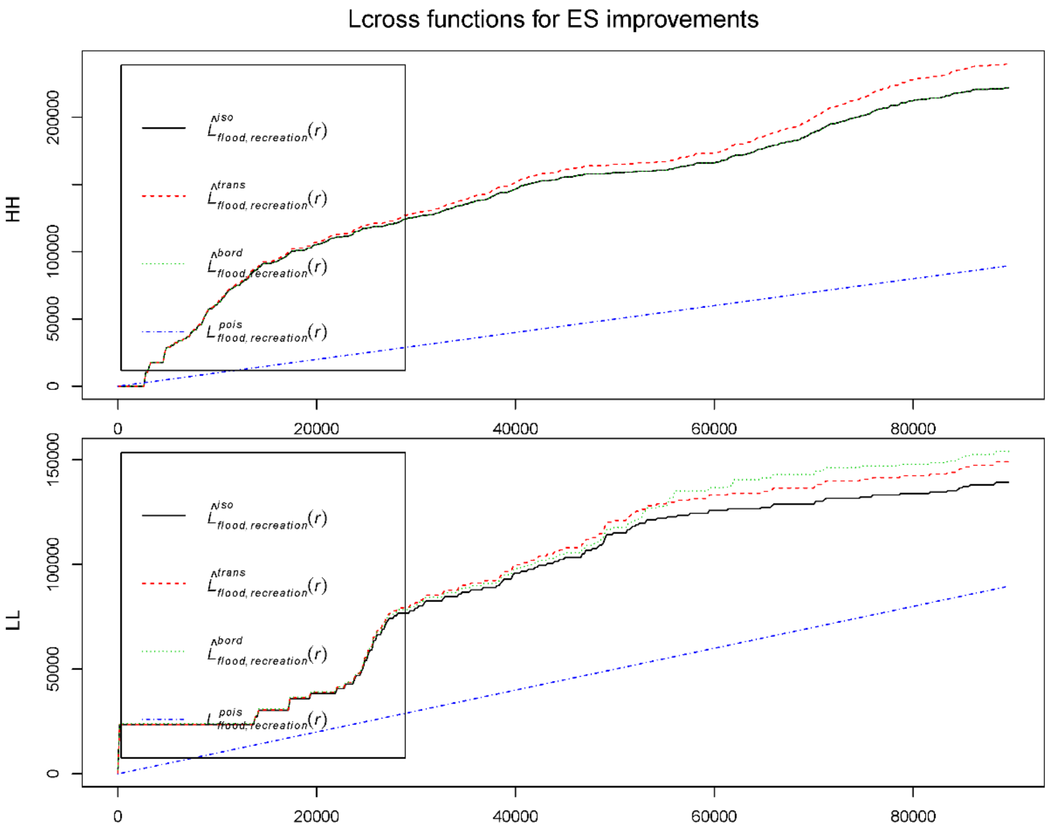

Lcross functions for surveys
HH
LL
0
50000
100000
150000
200000
20000
40000
60000
80000
$\hat{L}^{iso}_{Forth,Tay}(r)$
$\hat{L}^{trans}_{Forth,Tay}(r)$
$\hat{L}^{bord}_{Forth,Tay}(r)$
$L^{pois}_{Forth,Tay}(r)$

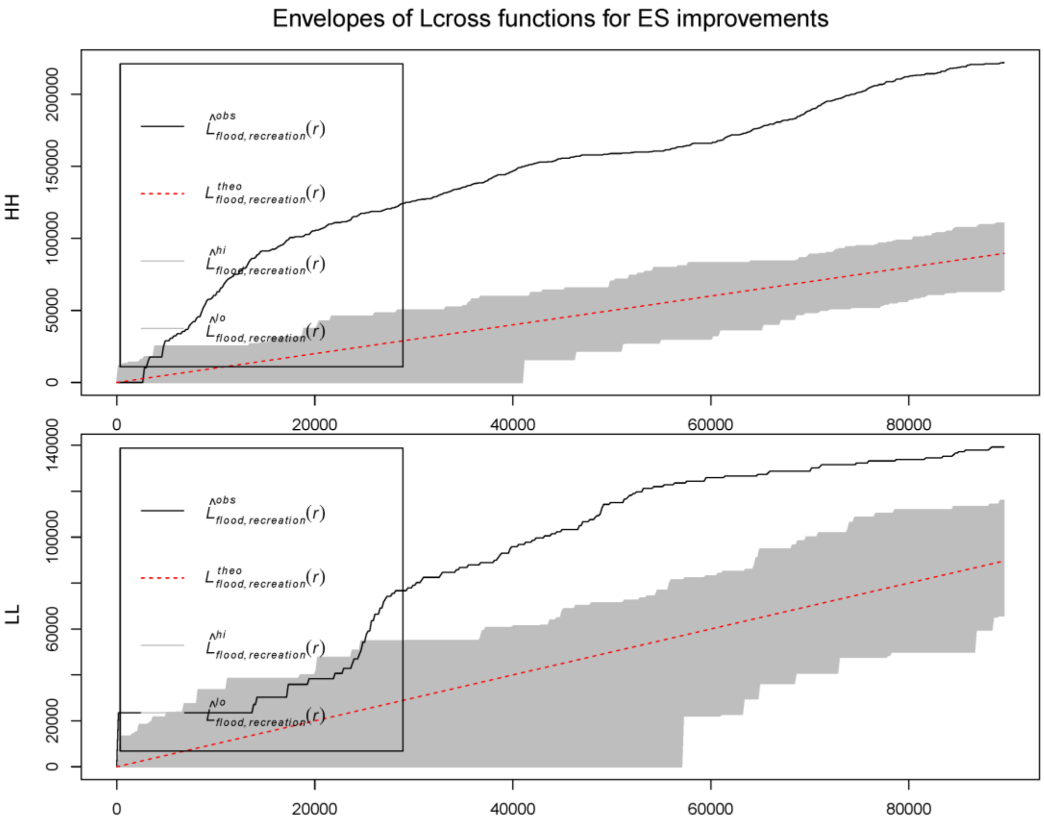

Envelopes of Lcross functions for surveys
HH
LL
0
50000
100000
150000
200000
20000
40000
60000
80000
$\hat{L}^{obs}_{Forth,Tay}(r)$
$L^{theo}_{Forth,Tay}(r)$
$\hat{L}^{hi}_{Forth,Tay}(r)$
$\hat{L}^{lo}_{Forth,Tay}(r)$

Annex 25 L-cross functions and envelopes for local clusters of WTP estimates marked by ES

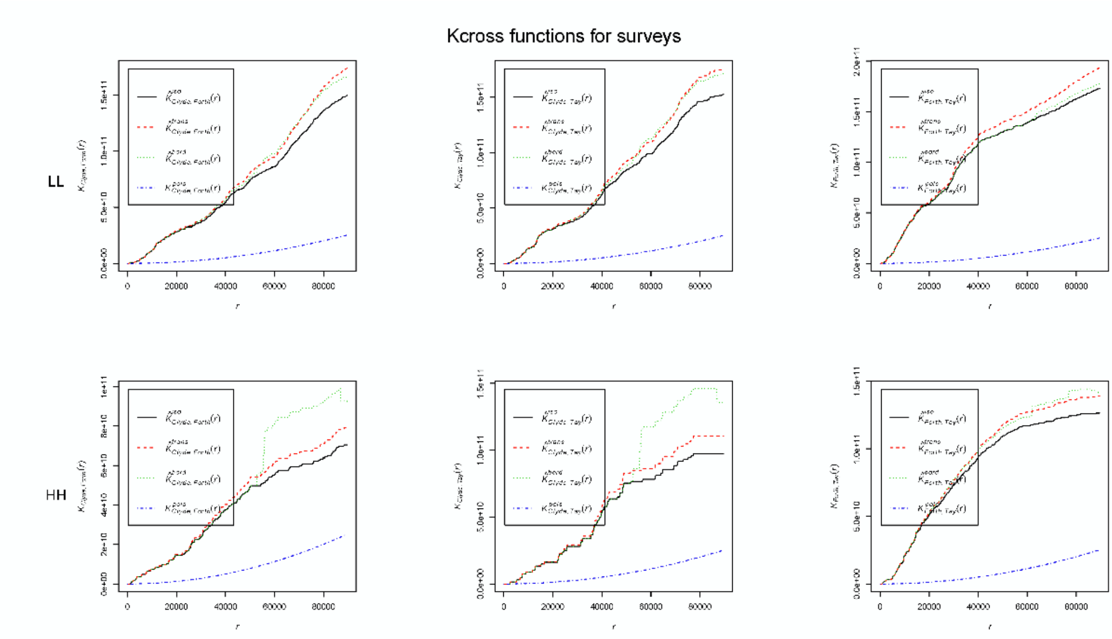


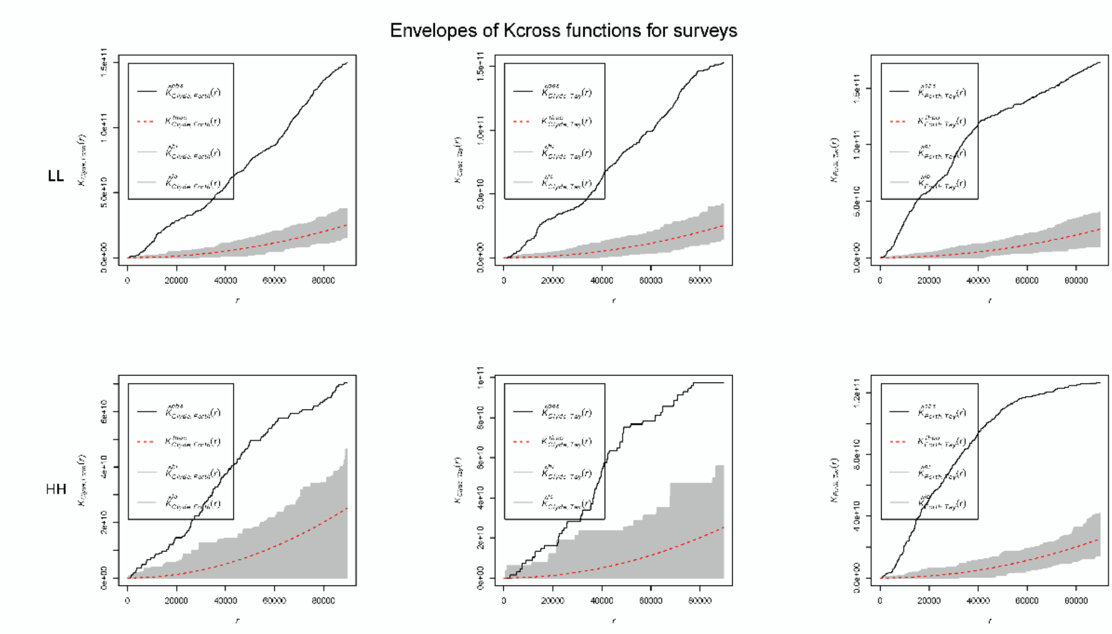



Lcross functions for ES improvements
HH
LL
$\hat{L}^{iso}_{flood,recreation}(r)$
$\hat{L}^{trans}_{flood,recreation}(r)$
$\hat{L}^{bord}_{flood,recreation}(r)$
$L^{pois}_{flood,recreation}(r)$
0
20000
40000
60000
80000
50000
100000
150000
200000

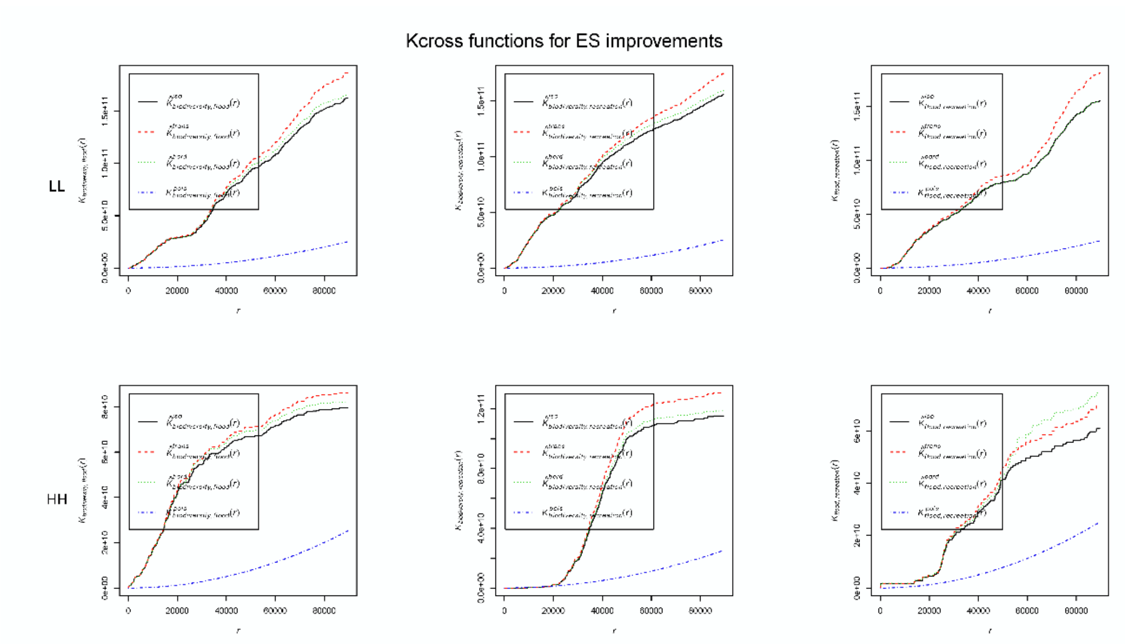

Envelopes of Lcross functions for ES improvements
HH
LL
$\hat{L}^{obs}_{flood,recreation}(r)$
$L^{theo}_{flood,recreation}(r)$
$\hat{L}^{hi}_{flood,recreation}(r)$
$\hat{L}^{lo}_{flood,recreation}(r)$
0
20000
40000
60000
80000
50000
100000
150000
200000
60000
140000

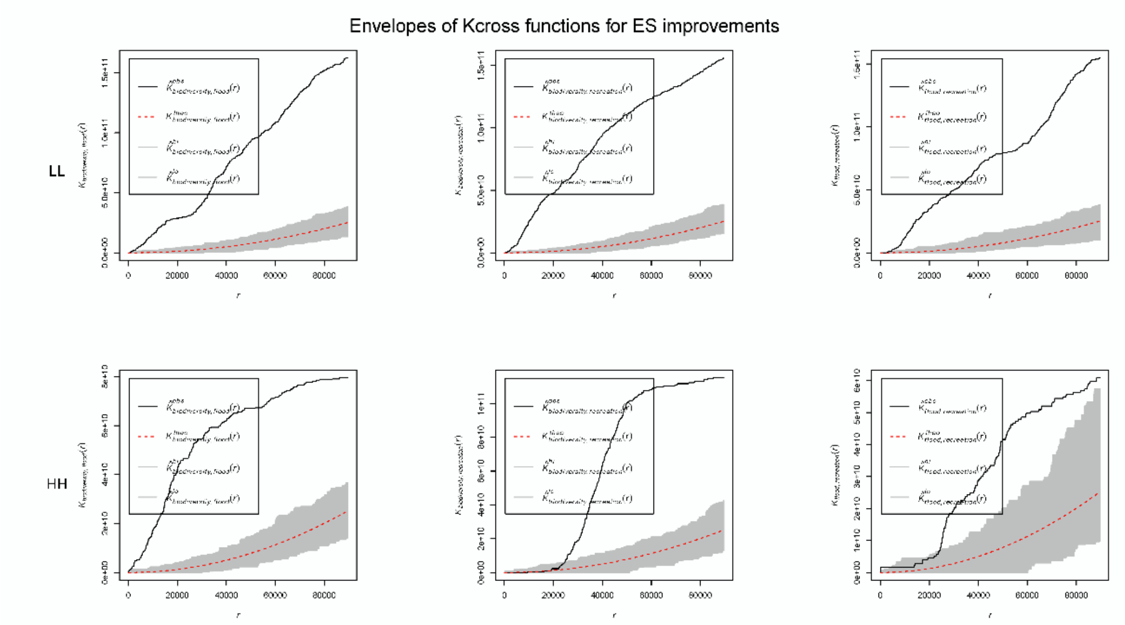
Annex 26 K-cross functions and envelopes for local clusters of WTP estimates marked by survey





Annex 27 K-cross functions and envelopes for local clusters of WTP estimates marked by ES





Annex 28 Summary statistics of respondents and their households (n=473)

Variable	Mean	S.D.
Income (net, in £ per month)	1852.16	1142.96
Age	50.29	16.32
Household size	2.37	1.25
Gender (% female)	53.07	
Education (% with university degree and above)	39.75	
Employment (% economically active)	62.37	
Residency in the area (% residents)	30.66	
Visited the area for outdoor recreational activities (% visitors)	54.12	
People perceiving a better environmental status in the area than 10 years ago (% respondents)	19.87	
People perceiving a worse environmental status in the area than 10 years ago (% respondents)	20.72	

Source: Scottish estuarine management Choice Experiment, 2016.

Annex 29 RPL estimates for ES improvements

	RPL					
Attribute	Coeff. (Mean)		S.E.	Coeff. (S.D.)		S.E.
F1	1.60	***	0.11	0.67	*	0.15
F2	2.03	***	0.14	1.21		0.14
B1	1.66	***	0.13	0.34	***	0.23
B2	1.82	***	0.14	0.82	***	0.13
R1	0.63	***	0.09	0.07	***	0.23
R2	0.63	***	0.09	0.59	**	0.14
Cost	-0.02	***	0.00	-	-	-
ASC	-1.75	***	0.30	3.02	***	0.28
Log-likelihood	-2230.64					
Observations	2838.00					
Adjusted rho-sq	0.28					
AIC	4491.28					
BIC	4580.54					

Two-tailed t-test indicate values approaching close to significance (+) and with 10% (*), 5% (**) and 1% (***) significance levels. Standard errors computed by Delta method.